Vegetation Management Plan:
California Department of Fish and Game
“Cunningham Marsh” Conservation Easement Site, Sonoma County, California

Prepared for
California Native Plant Society, Milo Baker Chapter

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Executive Summary

Cunningham Marsh, like Pitkin Marsh, is an isolated wetland that historically supported a rich and distinctive regional wetland flora, including endangered plant species and many disjunct populations of plants typical of northern bog-like habitats. The California Department of Fish and Game holds a conservation easement on a 19-acre riparian wetland tributary to Blucher (syn. “Bloucher”) Creek. The easement site is dedicated to the protection of the endangered Pitkin Marsh lily, which survives at the edge of riparian wetlands. The riparian wetlands and associated oak savannah and grasslands have been altered by over a century of agricultural land use, and now also occur among rural residential developments. Major restoration of historic wetlands is probably not currently feasible because of site constraints and scientific constraints. Instead, feasible vegetation management and small-scale experimental restoration measures are proposed for existing plant communities. Pitkin Marsh lily management includes creating larger gaps in willow canopies; managing deer exclosures to restrict severe herbivory impacts; removal of Himalayan blackberry thickets, and replacement with native rush-sedge and riparian scrub vegetation; experimentation with “nurse shrubs” and vegetation gaps to partially protect clones and facilitate seedling emergence and establishment; and establishing new populations. Old non-native Monterey pines are proposed for conversion to wildlife trees (standing snags), while native oak saplings and seedlings would be transplanted and protected. White poplar colonies are proposed for conversion to willow and rush-sedge marsh. Rush-sedge-rush marsh within the channel floodplain may be diversified by excavating depressions or troughs in accreted floodplain sediment. Grasslands would be managed to reduce non-native annual grasses by several years of seasonally timed repeated mowing and haying, followed by maintenance mowing /haying in summer, or mowing with occasional fall burning or spring grazing. Guidelines for establishment of native perennial sod-forming grasses and sedges of riparian edges are included. Species-specific management strategies are included for a number of species of concern. Long-term management of the site should include many field trials and practical experiments that apply combinations of vegetation management techniques, along with continuity of careful observation, and long-term inspection or monitoring of results. For restoration and management purposes, long-term continuity of observations and documentation is more important than short-term, intensive data collection and analysis.
1.0 Background, Scope and Purpose

1.1. Background: Summary of Conservation history at Cunningham Marsh

Cunningham Marsh is one of three historic perennial wetlands in the Sebastopol area, southern Sonoma County, with similar distinctive local wetland floras, including numerous endemic or disjunct species. These distinctive Sonoma County wetlands have ecological and floristic affinities with acidic, nutrient-poor sandy/peaty fens, carrs, and bogs (oligotrophic, or nutrient-poor, wetlands) occurring mostly in sandy coastal terraces of the northern California and Oregon coast and northward. Unlike the Santa Rosa vernal pool wetlands, which are desiccated in summer and have limited influence by groundwater and streamflow, Cunningham Marsh is influenced by groundwater year-round, and is dependent on streamflow for flooding in winter and spring.

The California Department of Fish and Game (CDFG) holds a conservation easement on portions of privately owned parcels near the end of Big Cedar Road, including a tributary stream of Blucher Creek that discharges to the main Cunningham Marsh (Blucher Creek floodplain). The 19-acre CDFG easement site includes riparian wetlands of this arm of Cunningham Marsh, and includes multiple colonies of Pitkin Marsh lily, currently treated as a rare subspecies of the widespread leopard lily (*Lilium pardalinum* ssp. *pitkinense*). This plan focuses on the long-term vegetation management of this site, with emphasis on the long-term conservation of the Pitkin Marsh lily population.

Cunningham Marsh is one of three perennial wetland complexes in southern Sonoma County that historically supported many disjunct (wide gaps in geographic range), endemic (narrow and local geographic distribution), and regionally rare wetland plants. Pitkin Marsh and Perry Marsh have distinctive local floras sharing many species with Cunningham Marsh (Best *et al.* 1996). Pitkin Marsh has the richest native wetland flora, and the highest number of co-occurring disjunct species typically found in acidic, nutrient-poor wetlands of the northern Sonoma and Mendocino outer Coast Ranges (Table 1). These three “Marshes” (in the broad sense; wetlands including emergent marsh communities and other types of wetlands) have been interpreted as remnant floristic “islands”, relics of past climates and plant migrations (Rutzbzoff 1953, Best *et al.* 1996).

Cunningham Marsh is highly ranked among natural areas in Sonoma County for its high biological conservation values. Cunningham Marsh and Pitkin Marsh are the only natural localities of the endangered (Federal, State) Pitkin Marsh lily, *Lilium pardalinum* ssp. *pitkinense*. Cunningham Marsh overall has historically supported ten rare and endangered plant species, three of them federally listed, and one federally listed invertebrate species. Many have apparently become extirpated, but some may persist as dormant seed banks that could potentially regenerate mature populations. It is not known what proportion of these rare plant populations of past surveys occurred within the current CDFG easement site.
In 1998, the Cunningham Marsh Conservation Easement was established to protect marsh habitat and uplands from a proposed subdivision development on adjacent land. Under the easement agreement, the Department of Fish and Game (DFG) has jurisdiction over preservation of the 19-acre site; the Milo Baker Chapter of CNPS cooperatively assumed responsibility for monitoring and maintaining the Pitkin Marsh lily population within the easement. The easement eliminated excessive cattle grazing, with the intent of enabling recovery of rare native plants. CNPS site stewards have reported that since cattle were removed, some invasive non-native plants (particularly Himalayan blackberry, *Rubus discolor*) have increased significantly in abundance, and threaten to dominate much of the riparian wetland.

Current stewardship of the site by CNPS-Milo Baker Chapter includes:

- Installation and maintenance of wire cage exclosure (deer/horse fencing) around both clonal and newly detected seedling populations of Pitkin Marsh lily populations, to prevent browsing damage by overabundant deer;
- Removal (reduction) of Himalayan blackberry and other invasive non-native plants from near and within the fenced lily populations.
- Biennial lily census work (population counts).

CNPS has also proposed additional vegetation management at Cunningham Marsh. Previous suggestions for managing the site have included:

- An overall strategy for weed control and rare plant monitoring;
- Repair of livestock-degraded riparian corridor that provides needed water to the marsh well into the summer;
- Restriction of woodcutting, trash or yard waste dumping, plant collecting, burning.
- Habitat restoration to include replanting of native wetland plants that are now absent or scarce. Plants suggested for reintroduction by CNPS members include confirmed historic Cunningham Marsh species, as well as some rare plants from similar wetlands in the region or Pitkin Marsh endemics (beaked-rush, Sebastopol meadowfoam, Gairdner’s yampah, California bellflower, white sedge, and Pitkin Marsh paintbrush).

1.2. Purpose of Vegetation Management Plan

The basic goal of the vegetation management plan (VMP) is to sustain long-term viability of native plant populations within local dynamic (i.e., unstable) plant communities, representing the natural vegetation composition, dynamics and structure of Cunningham Marsh and its historic flora to the greatest extent feasible. The conservation of the local Pitkin Marsh lily population is among the primary objectives of the plan. The conservation goals of the plan are evidently constrained by the historic agricultural and residential land use changes imposed on Cunningham Marsh itself, and its tributary streams and watershed. The purpose of the plan assumes a goal of enhancement or modification of existing native plant...
communities and populations, rather than a major ecosystem restoration (full restoration or re-construction of natural vegetation, soils, and ecological processes. A full “restoration” of natural, historic pre-agricultural Cunningham Marsh conditions is constrained because of:

(1) The CDFG easement site includes only a small portion of historic Cunningham Marsh and its watershed;

(2) minimal information on pre-agricultural ecological conditions of local wetlands, and relatively high uncertainty of major restoration outcomes;

(3) substantial historic changes and constraints in basic hydrologic, soil and vegetation conditions, and adjacent land uses;

(4) limited potential for suitable source populations of locally extinct endemic plants for reintroductions;

(5) low feasibility, and high impacts and cost of full restoration within existing land ownership and adjacent land uses.

The site is surrounded by rural residences, and has limited access for major restoration work involving heavy equipment or earthmoving. There is currently no long-term baseline study of the site’s hydrology (seasonal variability in groundwater levels, groundwater quality, streamflow, duration and pattern of flooding and soil saturation) to guide a major restoration of freshwater marsh compatible with the full range of historic Cunningham Marsh plant species. It is also not clear to what extent the riparian wetlands of this arm of Cunningham Marsh included typical features of the main marsh, or the historic Cunningham Marsh conditions.

Paleoecological research on Cunningham marsh in the future may be used to inform more ambitious restoration designs. Data on buried soil horizons (plant macrofossils, peat composition, sediment stratigraphy), microfossils (pollen stratigraphy) within the larger Cunningham Marsh would be useful in estimating basic restoration parameters such as seasonal and interannual variability of hydroperiods, nutrient availability, and species composition changes over time and changing climates. Adequate sampling for such research would likely require access over much of the marsh, beyond the current CDFG easement boundaries. It would benefit CNPS and CDFG to actively inform wetland paleoecologists at universities in the region (e.g. Roger Byrne, U.C. Berkeley) that there are opportunities to sample sediment/peat cores at this rare wetland complex.

This VMP focuses on a suite of actions compatible with existing and likely future adjacent land uses, ranging from near-term to long-term activities. The plan proposes to continue and modify ongoing volunteer plant conservation work on-site, and proposes a rationale, strategy, and site-specific methods for long-term management of Cunningham Marsh’s distinctive
wetland vegetation. The plan also identifies key areas for potential research and on-site experimentation to expand the scope of future potential wetland management and restoration.

The objectives of this VMP for the CDFG easement site are:

- To conserve viable local endemic populations of Pitkin Marsh lily;
- To facilitate vegetation dynamics and structure that promotes long-term sexual and clonal reproduction of Pitkin Marsh lily;
- To promote the recovery of other local endemic populations of rare wetland plants (existing populations or re-emergent from dormant seed banks), if feasible;
- To gradually reduce and minimize (to the greatest extent feasible) the need for active, intensive, chronic maintenance of vegetation and lily populations, while promoting recovery of conserved rare or other native plant populations;
- To reduce spread of non-native plants, and avoid their local dominance in the wetland vegetation;
- To maintain or improve the wildlife habitat value of the conserved riparian wetlands, its ecotone, and upland grassland vegetation.

2.0 Historic and existing conditions: soils and vegetation of the project site

2.1. Existing and pre-agricultural soil conditions in the vicinity of Cunningham Marsh

The Soil Survey of Sonoma County (USDA 1972) provides basic soil descriptions for the stream valley including Cunningham Marsh area of lower Blucher (syn. “Bloucher”) Creek. The soil types in the vicinity of Cunningham Marsh drainage belong to the Goldridge and Blucher series. They are derived from fine sandy, silty, or clayey marine sediments associated with the Merced or Ohlson Ranch formations.

The stream valley and floodplain of Blucher Creek (syn. “Bloucher” Creek) itself are associated with Blucher series soils, and the majority of Cunningham Marsh. They occur within very gently sloping alluvial basins along stream bottoms, or on alluvial fans, at elevations below 300 feet (Figure 1). They consist of poorly drained loam over alluvium composed of stratified silt and clay. Typical soil profiles of the Blucher series are described as
“surface layer dark gray, medium acid and neutral loam and silt loam about 20 inches thick” above “a layer of gray, moderately alkaline fine sandy loam”. At depths between 34 to 60 inches, various darker gray heavy clay loams occur; these are associated with shallow water deposition in low-energy flow conditions. Blucher fine sandy loam occurs, distinguished by alluvial deposition of fine sandy loam, stratified by thin layers of light clay loam; these indicate past episodes of high-energy flood events alternating with calm water deposits. Buried A horizons occur, also indicating historic episodes of flooding and sedimentation from soil erosion in the agricultural watershed. Soil reaction of the A horizon in historic times is described as “medium acid to moderately alkaline”. Mottles are distinct and prominent, indicating prolonged waterlogged conditions. Permeability and runoff are slow. Effective rooting depth to the water table is a maximum of 40 to 50 inches. (USDA 1972).

The CDFG easement site is located on a small tributary stream of Blucher Creek (Figure 1). Soil series dominant in the local watershed are mapped as Goldridge series soils, with some Sebastopol series. These are fine sandy loams associated with uplifted marine terraces. Sebastopol sandy loams have clay subsoils that may restrict subsurface drainage. At a small scale, inclusions of Sebastopol soils may occur within areas mapped as Goldridge, and vice versa. Sebastopol fine sandy loams of the central riparian corridor are strongly acidic to very strongly acidic (pH 5.0 to 5.2 for Goldridge sandy loams; to pH 4.3 for Sebastopol sandy loams), in contrast with Blucher loams of the alluvial zones, which are moderately acidic to slightly alkaline (USDA 1972). Restricted soil drainage, acidic soil reaction, and hydrology of the stream corridor were conducive to development of nutrient-poor wetlands.

The unnamed tributary is incised where subsoil horizons are often locally exposed, or near the surface, due to past erosion: leached, white, fine, well-sorted sands somewhat similar to those occurring on Mendocino coastal terraces (but here lacking hardpans) known for fens and seasonally wet, “pygmy forest” Blacklock soil series (also nutrient-depleted, highly acidic, sandy). The acidophilic (acid soil-loving) historic flora of Cunningham and Pitkin Marsh are more likely to be associated with wetlands derived from these leached sands than from alluvial deposits of floodplains.

It appears likely that the conversion of the watershed to grazing and cropland (Figure 2) included periods of overgrazing and soil destabilization, resulting in significant sediment deposition during extreme flood events. This interpretation is consistent with the description of buried (single or multiple) A horizons, and description of soils with eroded upper horizons (USDA 1972). While episodes of catastrophic sediment deposition could occur in natural conditions, upper buried soil horizons are likely to be associated with historic agriculture. Pre-agricultural wetland plant communities included oligotrophic wetlands, formed on acidic, nutrient-poor sandy soils with sedge peats or swamp (woody wetland scrub), similar to fens and wet coastal grasslands of the north coast marine terraces in Mendocino and northwestern Sonoma Counties. Some of the buried A horizons described in the Blucher soil series may be associated with former peats of marshes or fens. Historic deposits of nutrient-enriched sediments from erosion of agricultural watersheds are likely to have fundamentally
altered the soil properties and vegetation dynamics of Cunningham Marsh.

Increased soil fertility and disturbance of historic vegetation may pose persistent problems for competition between naturally slow-growing, stress-tolerant native plants (adapted to nutrient-poor hydric soils) and non-native invasive ruderal plants with high relative growth rates. Historic sedimentation, disturbance, nutrient enrichment, and alteration of hydrology are likely to favor generalist wetland species with relatively high growth rates and strong colonizing ability and competitive ability. Slow-growing, slow-dispersing, stress-tolerant wetland plants are likely to be reduced to relict or declining populations when old, stable, unproductive wetlands are disturbed and made more productive. This appears to be relevant to the current status of wetlands in and around Cunningham Marsh as well: most of the vegetation appears young and consists of widespread species, but a significant number of uncommon or regionally endemic plants persist, or have persisted until recent years.

2.2. Historic flora and vegetation of Cunningham Marsh.

There are no available botanical descriptions of pre-agricultural Cunningham Marsh vegetation that could assist development of adequate ecological models for restoration of natural, native wetland vegetation. Soil series descriptions dating back to early agricultural conversion state only generally that “the original cover of grasses, sedges, and vines has been modified by use for cultivated crops or pasture” (USDA 1972). The biogeographic patterns of Pitkin Marsh, Perry Marsh, and Cunningham Marsh floras in southern Sonoma County were analyzed by Rubtzoff (1953). Pitkin Marsh shares much of the distinctive historic flora of Cunningham Marsh. It includes a somewhat richer local native wetland flora, but both including many rare or disjunct populations of marsh plants associated with bogs, fens, and marshes typical of acidic, nutrient-poor (oligotrophic) north coast wetlands. These distinctive, isolated southern Sonoma wetland floras are interpreted as relics of past climates and plant distributions. Older herbarium collections from the region, unfortunately lacking modern specific locality information, also include northern bog species such as *Eriophorum gracile* (cotton-sedge, often today associated with *Rhynchospora* spp. in bogs or fens) from “swamps near Santa Rosa” (Bigelow collection in 1854, Best *et al.* 1996). Plants indicative of oligotrophic wetlands (bogs, acidic fens) that have been reported from the Pitkin/Perry/Cunningham wetlands are distinguished in Table 1. These wetlands had somewhat similar, but distinct floras, and probably distinct vegetation.

Pitkin Marsh also has been reduced, isolated, and degraded to a condition that would also have limited value as a reference site or ecological model for restoration of Cunningham Marsh wetlands, even if they had been equivalent. A 1953 study by Peter Rubtzoff described the plant species then present in Pitkin Marsh as well as in two smaller marshes in the same region, Perry Marsh and Cunningham Marsh. The Rubtzoff study referred to collections made by Milo Baker between 1934 and 1952 and by John Thomas Howell in 1946-1947. Rubtzoff included Perry and Cunningham marshes in his studies because of their floristic
similarity to one another, their geographic isolation from floristically comparable wetlands, and their local geographic proximity.

The name “Cunningham Marsh” was probably applied during the historic agricultural period during or after the period when dairy and ranch operations converted the original wetland vegetation. Portions of the Blucher Creek floodplain used as cattle pasture today (downstream from the CDFG easement site) do support true freshwater marsh vegetation. But much or some of the wetland complex in the vicinity of historic Cunningham Marsh historically included a number of plant species that have high fidelity to acidic, nutrient-poor wetland types such as bogs and fens with much woody vegetation, consistent with the description of “swamps near Santa Rosa”. The prevalence of at least slow-flowing water (Blucher Creek) through sandy soils on gentle gradients (USDA 1972), lack of deep peat, and distinctive historic floristic composition indicates that the original “marsh” was neither a bog nor typical freshwater marsh, but perennial wetland with fen or carr and marsh characteristics. Fens are characterized by a mix of herbaceous plants in nutrient-poor, acidic sandy or peaty soils with very slow-flowing water; carrs are similar, but shrub-dominated.

The geographic boundaries of past survey areas of “Cunningham Marsh”, and their relation to the boundaries of the current CDFG easement site within “Cunningham Estates”, are unclear. It appears that the largest areas of open marsh (emergent cattail, sedge, rush assemblage) lie downstream of the CDFG easement site. The riparian wetlands of the CDFG easement site itself are dominated by riparian willow scrub and blackberry thicket, with pockets of sedge/rush marsh and a perennial creek with low summer baseflow. As a small subsample of Cunningham Marsh with and primarily riparian wetland vegetation rather than an emergent marsh basin, it is reasonable to assume that the CDFG easement site would naturally capture a relatively small and biased sample of the entire Cunningham Marsh flora and plant associations.

A 1990 survey of the current CDFG easement site by Betty Guggolz indicated the presence of several sensitive plants, including Sebastopol meadowfoam (Limnanthes vinculans), round-headed beaked rush (Rhynchospora globularis), Sonoma alopecurus (Alopecurus aequalis var. sonomensis), and a cinquefoil identified as Hickman’s cinquefoil (Potentilla hickmanii; but see section 5.2.13). None of these species could be found in subsequent surveys, in 1993 and 1994.

### 2.3 Overview of existing vegetation of the CDFG easement site

The following general description of vegetation at the CDFG easement site is based on three site visits by the author (May and September 2004, April 2005), and reference to the composite “Cunningham Marsh Plant Species List” maintained by the Milo Baker chapter of CNPS, updated 12/8/03. The purpose of the site visits was to observe overall patterns of soil, vegetation, plant species distributions, topography, and hydrologic indicators relevant to
management and restoration of the site. No new floristic surveys were conducted.

The CNPS species list is a compilation of plant species identified on the Chilton/Voit property (Cunningham Estates) in 1993-4, site-specific observations by CDFG (Gene Cooley) and knowledgeable CNPS members, and earlier “Cunningham Marsh” reports by regional expert botanists (Milo Baker, Nancy Harrison, Charles Quibell, Peter Rubtzoff) and herbarium records or localities reported in A Flora of Sonoma County (Best et al. 1996). The consistency of geographic scope in past surveys compiled in this list is unclear, and it should be interpreted with caution. Some plants reported from “Cunningham Marsh” may be well outside the boundaries of the CDFG easement. Some past Cunningham Marsh plant identifications are either taxonomically problematic (due to taxonomic revisions or ambiguities; e.g. Potentilla hickmanii), or may be simply erroneous (misidentifications, key errors; e.g. Juncus acutus, Pinus muricata).

The CDFG easement site includes multiple vegetation types across an elevation gradient between the stream channel and the gently sloping uplands. The gradient includes grassland communities, stands of native oaks, non-native pines and eucalyptus, and a stream channel and active floodplain associated with riparian woodland/scrub and freshwater marsh.

The upper slopes of the CDFG easement site in summer and fall are dominated by mixed non-native grassland (apparently former pasture), with very few (but significant) oak seedlings (Figure 3). The detailed composition of the grassland was not investigated, but the CNPS species list indicates the presence of native elements such as blue-eyed grass (Sisyrinchium bellum), native perennial grasses (Elymus glaucus, Danthonia californica, Deschampsia cespitosa, Leymus triticoides, Nassella pulchra, others) and numerous bulb-bearing taxa (Brodiaea, Calochortus, Tritelia, etc.) species. For this reason, “non-native grassland” should not be interpreted to be lacking in native grassland species throughout the year. In April 2004, however, non-native grasses were strongly dominant, and native forbs and bulbs appeared to be scarce in grasslands. Frequent occurrence of the non-native sheep-sorrel (Rumex acetosella) suggests naturally acidic sandy soil. Non-native grasses appear to include a mix of common annual species and some perennial cespitose forage grasses (possibly erect terrestrial ecotypes of Agrostis stolonifera (redtop) or A. capillaris). There is no documented information on the relative abundance or distribution of the native vernal grassland flora reported in the CNPS species list for the site. Multiple seasonal survey times (vernal flora, summer graminoids) with notation of rank abundance of species would be needed to adequately characterize the condition of the grassland for management (see Section 6.4).

Scattered large, mature oaks (Quercus agrifolia, Q. kelloggi, Q. lobata, Q. garryana) overlap with grasslands and the riparian wetlands, primarily at lower elevations near the transition to the riparian scrub (described as “mesic forest” habitat in one CNPS species list). Oaks provide much of the upper canopy later in riparian woodland. Large, mature to decadent (and some dead) Monterey pines (Pinus radiata; apparently reported as P. muricata, but specimens I
observed had cones lacking prickles on scales, and leaves are in 3’s or 2’s, consistent only with *P. radiata*) occur congested among oaks near the transition zone with riparian woodland (Figure 4). The pines were apparently planted as windbreaks many decades ago (likely > 50-70 yr, possibly > 100 yr), because there is relatively little variation in size or age structure of the pine stand. It is likely that the pines were planted along with the few, old blue gum trees (*Eucalyptus globulus*) at the upstream end of the site. Monterey pine and blue gum are common historic agricultural windbreak tree plantings near the coast. The pine-oak stands have either grassland or shrub thickets as understories. Decadent pines and snags (standing dead trees) are either roost or nest sites for great horned owls, as indicated by large pellets found beneath them (September 2004). Because of the land use history and the small scale of the site, the assemblage of native oaks and non-native windbreak tree relics are appropriately interpreted as stands rather than a formal series or association.

The transition zone (ecotone) between well-drained upland grasslands and the riparian vegetation occurs where near-surface soil moisture persists longer into the growing season. Soil moisture in spring appears to be related to discharge of groundwater (surface seeps, saturated soils) in the riparian zone. At the ground layer and shrub layer, this transition zone does support a distinct natural vegetation, despite abundance of some non-native species. This zone is indicated by increased frequency and size of shrubs and small trees, including blackberries (*Rubus ursinus, R. discolor*), dogwood (*Cornus sericea*), hawthorn (*Crataegus douglasii*), sedge (*Carex barbarae*), bracken (*Pteridium aquilinum*) and some mesic grasses such as velvetgrass (*Holcus lanatus*). This zone also includes some seasonal or ephemeral stream channels that expose underlying whitish sandy subsoil. Some depressions in gullies (seasonal or ephemeral minor drainages) develop local patches of mesic/wetland vegetation (*Carex obnupta, Rhododendron occidentale, Rubus ursinus*). Some Pitkin Marsh lily populations occur at the lower portion of this ecotone zone. Oaks provide much of the upper canopy cover in the transition zone.

The riparian ecotone (Figure 5) grades into true riparian wetlands (upper soil horizons soil saturated for weeks or months). The upper riparian wetlands consist of a woodland/thicket zone (carr) is marked by increased density of native and non-native blackberries (*Rubus ursinus, R. discolor*), and frequent occurrence of western azalea (*Rhododendron occidentale*), soft rush (*Juncus effusus*), often intermixed. Oaks remain abundant to dominant in the overstory layer. Sedges (*Carex* spp.) are locally common, and non-native velvetgrass (*Holcus lanatus*) is common to abundant. California wax-myrtle (*Myrica californica*) occurs in the upper riparian zone, and in lower, wetter riparian woodland as well. The soil in this zone consists of a thick layer of duff (leaf litter) and dark, fibrous organic upper soil horizons, above sand or sandy loam. Most of the Pitkin Marsh lily (*Lilium pardalinum ssp. pitkinense*) colonies occur in this zone (Figure 6).

The lower riparian woodland/thicket is marked by dominance of tall arroyo willow thickets (*Salix lasiolepis*) with native and non-native blackberry subdominant, and a sparse ground layer. The lower riparian woodland/thicket exhibits soil and surface indicators of persistent
soil saturation or flooding in the winter and spring, and includes the stream channel with some standing water in summer. Within the lower riparian zone, and in some drainages leading to the main stream channel, true freshwater marsh patches occur. Freshwater marsh also occurs immediately above and below a concrete culvert in the stream. The marshes are variable dominated by soft rush (*Juncus effusus*), but also include Mexican rush (*J. mexicanus*), Carex spp., and water-parsley, *Oenanthe sarmentosa*. Below the stream culvert, freshwater marsh includes substantial stands of *Scirpus microcarpus* and *Carex obnupta* as well, with *Urtica dioica* locally abundant. A small clonal grove of European white poplar (*Populus alba*) occurs below this isolated marsh patch. Downstream, the riparian wetlands connect with the larger Cunningham Marsh (offsite), currently dominated by a mix of woody and emergent herbaceous and graminoid wetland plants, predominantly *Juncus effusus*, *Carex obnupta*, *Rubus* spp., *Typha* sp. and *Urtica dioica*. The marsh does not appear to have recently supported marsh vegetation typical of inundation for long periods during the growing season.

The presence of the concrete culvert in the channel, and the clonal grove of white popular, indicates that the riparian wetland vegetation was probably highly altered many decades (or longer) ago, probably about the time of the non-native tree windbreak plantings. The riparian woodland itself lacks large willows, and the marsh and thicket appear to be regenerated or partially regenerated, rather than remnants of original vegetation. Willows are generally less than 5 m tall, with dense branching and foliage near ground level. The mature oaks indicate that some old or original vegetation has persisted.

### 3.0 Plant Species of Concern

The Pitkin Marsh lily population at the CDFG easement site is the preeminent plant species of concern because it is one of two known populations, and the most protected against known threats. Its priority for management is also indicated by its federally endangered status. Other federally listed species have been identified on the site (or in Cunningham Marsh) in the past, but have not been observed since 1990. These include Sebastopol meadowfoam (*Limnanthes vinculans*), Sonoma alopecurus (*Alopecurus aequalis var. sonomensis*), and a cinquefoil identified as Hickman’s cinquefoil (*Potentilla hickmanii*). Other species found in the past on or near the site include some of the regionally rare disjunct species shared with the Pitkin Marsh flora, such as round-headed beaked rush (*Rhynchospora globularis*) and Bolander's reed-grass (*Calamagrostis bolanderi*).

Some CNPS members have proposed the site for establishment of additional conservation populations of some rare or disjunct plant populations from Pitkin Marsh (perhaps as a surrogate locality in case of failure to adequately protect Pitkin Marsh), but which have no known history or habitat at Cunningham marsh. These “Pitkin-only” introduction candidates might include include *Carex albida* (federally endangered) *Carex gynodynamis*, *Carex*
*lanuginosa*, *Carex nebraskensis*, *Gentiana sceptrum*, *Sparganium erectum*, and *Drosera rotundifolia*. In addition, the site includes a significant area of grassland (derelict pasture), sedge-rush freshwater marsh, willow-dominated riparian wetlands, mature native oaks, and seasonal wetland scrub that include important native communities of widespread plant species, and a few uncommon to rare plants that are not entirely restricted to wetlands, such as *Perideridia gairdneri* ssp. *gairdneri*.

Species specific recommendations and conservation strategies for species of concern are provided in section 5.2. Recommendations for reintroduction or introduction policies, and their biological rationale, are provided in section 4.1.

**4.0 Vegetation models and methods for Cunningham Marsh**

**4.1. Vegetation and flora: conceptual model for Cunningham Marsh management**

The sandy, somewhat acid sandy soils present, and persistence of at least one ericaceous shrub species, western azalea (*Rhododendron occidentale*; Figure 6), may be interpreted as remnants of the former oligotrophic wetlands (acidic, sandy or peaty wet grassland, fen or carr) that are implied by the historic flora of Cunningham Marsh. Some plants in the historic Cunningham Marsh flora are narrowly associated with oligotrophic wetlands (Table 1). The current wetland vegetation includes a very different species composition and vegetation: presence of nettle and cattail, luxuriant growth of blackberry, willows, and invasion of wetlands by bull thistle, all suggest relatively more productive, fertile contemporary wetland soil conditions. It is likely that these conditions are associated with sedimentation, disturbance, and many decades of nutrient enrichment from agricultural runoff (cattle, orchards) and groundwater leached from residential areas.

Although the local floras of Pitkin, Perry, and Cunningham Marsh wetlands form a biogeographically distinct group (Rubitszoff 1953), they are not equivalent with one another. Pitkin Marsh flora is substantially richer in species of genera shared by the cluster of wetlands. Cunningham Marsh also historically supported some uncommon to rare taxa that have not been reported from Pitkin Marsh (Table 1). The freshwater marshes of western and northwestern Sonoma County (Duncan Mills Marsh, lower Russian River, Gualala Point), share some of the species at the southern “Pitkin-affinity” wetlands, but are not ecologically equivalent to the oligotrophic elements of the Pitkin-affinity flora. The marine terrace wetlands of Sea Ranch, Plantation, and Salt Point, with many elements of northern bogs and wet ericaceous scrub, may be more ecologically similar to pre-agricultural Cunningham Marsh than most freshwater marshes in the south County. Marine terrace wetlands, however, are geographically remote from Cunningham Marsh, and occur in coniferous forest openings within a marine climate zone; they are not ecologically equivalent to Cunningham Marsh’s setting.
These floristic relationships among regional marshes suggest that Pitkin Marsh should not be assumed uncritically to be a restoration or reintroduction model for either Cunningham Marsh as a whole, or the riparian corridor wetlands of the CDFG easement site in particular. Any use of reference wetlands for restoration planning of this disjunct and unique wetland should be limited to general relationships of soil, vegetation, and hydrology. Because Pitkin and Cunningham Marsh have been isolated for long periods (geologic time), it is reasonable to presume that rare plant populations should not be translocated between them to “reintroduce” species historically reported from Cunningham Marsh, but presumed to be extirpated. The relationship of the CDFG easement site to the general “Cunningham Marsh” of Howell, Baker, and Rubtzoff collection localities is not clear, and the site may not support the wetland types that naturally would support all of the “missing” Cunningham wetland flora. Similarly, it should be presumed that it would be inappropriate to “reintroduce” remote rare plant populations from northern or outer coastal wetlands to Cunningham Marsh for the sake of replacing missing names on its historic species lists. It would be equally inappropriate to “pack” Cunningham Marsh with artificially augmented native species richness by stocking it with translocated rare Pitkin Marsh plant populations. The essential biogeographic significance of the Cunningham and Pitkin rare plant populations is their isolated, disjunct nature – a legacy of idiosyncratic plant migrations patterns over geologic time, and the distinctive local wetland environments. Substituting these distinctive, isolated populations with others derived from different parts of species modern ranges (i.e. different segregates of past migrations) would corrupt the scientific and conservation value of the Cunningham Marsh plant community, and defeat the purpose of its protection.

Taxonomic ambiguities of local historic populations (Potentilla hickmanii, Lilium pardalinum; see sections 5.2.13, 5.2.8) also justify caution about reintroductions of unrelated populations based on taxonomic treatments of variable rigor and accuracy. This relatively restrictive approach to reintroduction is fitting for the isolated, disjunct nature of the Marsh and its rare wetland species historically present. A less restrictive reintroduction policy may apply to relatively widespread species from nearby localities, but there is probably relatively little need to do so. Augmentation of native grassland species diversity, for example, might involve reintroduction of relatively widespread taxa from Sebastopol area source populations.

4.2. Summary of relevant techniques and tools for local vegetation management

The following is an overview of vegetation management tactics and methods that are either already in use at the CDFG easement site, or specifically apply to one or more of the specific species management or community management strategies (Sections 5.0, 6.0.). Because single techniques or tools may apply to multiple species and communities, they are treated together here, and cited where relevant.
4.2.1. Deer browsing exclosures

Overabundant black-tail deer utilize the cover of the riparian corridor, and appear to concentrate browsing in the moist, woody riparian scrub thickets where clonal populations of Pitkin Marsh lily occur (see 5.2.8.). Because deer browsing can severely impact annual seed production of the lily population, and may also affect growth and survivorship of clonal populations, local physical barriers to enclose sensitive plants and exclude browsing by deer (“exclosures”) have been applied. These currently consist of (improvised) poultry mesh or horse fence cages and pens, framed by metal vineyard fence stakes (Figure 7). The permanent exclosures appear to protect other species from effects of browsing as well: oak seedling recruitment and sapling growth is conspicuously higher within some browsing exclosures.

Permanent exclosures are currently justified for protection of the few existing, natural lily populations. In the long-term, as population sizes increase, it may be appropriate to modify exclosure structures or management to allow for some degree of plant–herbivore interactions that do not risk extirpation of the rare plant population. Otherwise, if survivorship and growth, and pattern of distribution of new lily populations always depend primarily on erection of exclosures, the population is effectively in cultivation in a wild setting.

Permanent exclosures may also cause local vegetation stands to undergo long-term local succession that is detrimental to the target species: for example, if high densities of oaks or other woody species develop in the absence of all browsing within permanent exclosures, additional artificial vegetation maintenance may be necessary to compensate for loss of intermittent browsing.

Assuming permanent exclosures are used to provide insurance against extinction of the original local ‘core’ clonal colonies of Pitkin Marsh lily populations, the extent or duration of exclosure use for additional colonies should be critically evaluated. If some natural browsing is allowed some years, modification of exclosure use could include: (a) variable duration of exclosure and exclosure-free years for individual colonies once they reach a critical (resilient) size; (b) temporary use of exclosures during early clonal growth, or during maturation of potential “nurse plants” (partially sheltering shrubs like Rhododendron occidentale; see sections 4.2.3, 5.2.12). Exclosure designs that merely reduce the frequency of browsing (i.e. provide partial obstructions, but not absolute protection) may be considered for well-established new clonal colonies. Partial exclosure would partially compensate for artificially intensive deer browsing, but would not commit the lily population to equally artificial elimination of all browsing pressure. Fence materials that discourage or merely reduce frequency of browsing, such as simple wire and stake structures, or high-test fishing line or nylon mesh bird netting “fence” barriers tied to stakes or woody shrubs may provide partial restriction of deer access to established satellite colonies of lilies. If full exclosures are to be reconstructed, horse or deer fencing (heavier-gauge rectangular wire mesh) would be preferable over existing poultry mesh fencing: it would allow easier entry and maintenance of exclosure areas by site stewards and volunteer monitoring crews, and would be easier to maintain.
4.2.2. Selective thinning, restructuring of willow canopy

Local successional patterns of vegetation appear to include overgrowth of willow canopies over blackberry/rush vegetation, including some lily colonies. Overhanging willow canopies appear to reduce lily flower visibility to pollinators, and reduce seed set. Vigor of understory vegetation, including lily colonies, may decline as willow canopy density and cover increases. Pruning (heading back) gaps in the willow canopy, or clearing willows around the periphery of lily colonies is a short-term tactic to increase light penetration to the ground layer and shrub layer. Because willows resprout “sucker shoots” (juvenile, unbranched “whips”, or “water sprouts”) rapidly at points of injury, light pruning may actually increase canopy density within a growing season. Pruning may need to set regeneration points of willows back from edges of the vegetation gaps to be maintained. Some direct herbicide application (“painting” of cut stumps, frilled bark; section 4.2.5) to willows growing into lily populations may be justified on a case-by-case basis, where management by pruning is inefficient. Close proximity of large willow trunks and lily populations may justify local herbicide treatment of individual willows, if oak overstory shade is retained.

4.2.3. Experimental facilitation/nurse plant structures

Many of the Pitkin Marsh lily clonal colonies occur either free-standing (entire clone exposed to deer browsing, lodging or breakage by wind) or interspersed among shrubs like Rhododendron occidentale that have sparse, semi-open interiors and canopies. Mature lilies clones also occur growing through relatively dense mats of Rubus ursinus that would probably inhibit establishment of lily seedlings. The portions of clones within the interiors of large R. occidentale are potentially less accessible for deer browsing, creating “refuges” for portions of clones within them (Figure 8). Physical support and sheltering by shrubs has been identified as a potential facilitative interaction with the ecologically similar western lily (Lilium occidentale; Guerrant et al. 1999). Portions of lily clones beyond the edges of R. occidentale may be more exposed to severe browsing, but are also open to “foraging” for soil nutrients and light, which may benefit the physiologically integrated clone. One experimental approach to establishing new lily populations would be to initiate some new colonies under protection of exclosures, but close to potential “nurse plants” such as R. occidentale. Clonal growth and reproductive success (seed production) could be compared between exclosure-free lily/azalea complexes and free-standing lily colonies, and exclosure-protected colonies. If lily-azalea complexes allow for adequate lily clonal survival and reproduction without exclosures over multiple years (5-10), this co-planting technique may indicate an alternative approach to wire mesh exclosures. Because azaleas are very slow-growing, initial experiments may need to exploit established azaleas.

4.2.4. Creation of vegetation gaps or excavated substrate (subsoil exposures, wetland depressions)
Several erosional gullies and windthrow of old trees have exposed “raw” sandy subsoil (no organic staining, nearly white) near in the ecotone between grassland and riparian wetlands at the CDFG easement site. These gullies are closer to the seasonally variable groundwater, and are subject to ephemeral streamflow. In some older gullies, wetland or mesic native vegetation (Carex obnupta, Rhododendron occidentale, Rubus ursinus, Juncus effusus) have vigorously colonized the exposed mineral substrate, while prevalent non-native invasive species appear to be at a competitive disadvantage in the infertile sandy subsoil depressions. At the south end of the site, an large artificial basin has been excavated as a seasonal pond in the middle of the grassland (apparently for waterfowl), but removed from adjacent seed sources of native wetland shrubs and graminoids, it appears to be dominated by only sparse cover of non-native grasses and forbs. The windthrow-exposed sand is scarcely colonized. The gully appears to be the oldest of the vegetation gaps. In addition, previous mechanical clearing of a Himalayan blackberry stand with heavy equipment in recent years has resulted in recolonization by a mixed stand of Rubus ursinus, Juncus effusus, and Carex spp., potentially suitable for lily colonies (Betty Young, personal communication 2004).

These natural and artificial vegetation gaps suggest some potential to promote development of native wetland or riparian vegetation by creating vegetation gaps, or by combining vegetation removal with exposure of subsoil or depressional topography. Gap creation by manual or chemical removal of existing vegetation (see 3.2.5) without altering substrate or hydrology is the simplest method. Experimental vegetation gap creation should be applied first in areas of predominant non-native vegetation, and may be combined with seeding or planting of vegetation (3.2.9), soil conditioning (3.2.6), or both. Gap sizes may need to be based on what is practical for labor and equipment, but variable gap sizes would be advantageous. Variable gap sizes are most likely to increase the range of microenvironmental heterogeneity that may potentially increase diversity of vegetation. Gaps created in intact soils should be expected to stimulate germination and emergence of existing seed banks, which may contain either abundant non-native or native seeds. Gaps are also likely to serve as seedling nurseries for adjacent populations that disperse seed into them. Therefore, if gaps are created next to invasive plant populations, those stands should be treated (mown, herbicide-treated,etc.) prior to gap creation in order to reduce seed dispersal into the gap.

Excavating slight depressions or troughs in the ecotone between terrestrial and riparian wetland zones is likely to increase the extent of wetland patches with predominantly native vegetation. Excavation would be performed mechanically (backhoe, bulldozer or scraper), and would require access to the treatment sites. It is unclear whether the existing topography reflects past leveling for pasture, or whether it is naturally lacking in minor drainages. It is reasonably likely that leveling was performed to maximize pasture or orchard area, in which case excavation of depressions or troughs is likely to provide surrogates for natural drainage features. The spontaneous erosion of a gully in otherwise stable grassland suggests that the existing slope and drainage pattern is not in natural equilibrium, particularly since stabilizing vegetation cover is otherwise largely continuous.
On-site disposal of excavated subsoil from created depressions or troughs may be problematic. Placement of excavated subsoil material in gently sloping (1:10 to 1:5) mounds up to about 0.5 m in relief above adjacent elevations would minimize artificial topographic impacts of disposal. Location of disposal sites in areas of grassland dominated by non-native annual species would minimize disruption of important plant communities. Mounds may be planted with oaks or native grassland species.

4.2.5. Physical removal, herbicide treatment of non-native vegetation.

Manual removal of woody or herbaceous vegetation can be accomplished in local patches using volunteer crews. The most suitable time for removal is late summer to fall, when most non-target native plants are either entering or in dormancy, and are most likely to escape damage. Manual removal can include techniques that entirely uproot plants (tile spades, weed wrenches), sever them below the ground surface (mattocks), or sever them at or above surface (mattocks, pruning saws, chain saws for woody plants; manual weed-whips/grass-whips or motorized weed-whackers for herbaceous plants or grasses). Herbicides compatible with sensitive wildlife or wetland areas may be used in combination with manual removal (to poison stump-sprouting) or independently. Glyphosate is widely used in wildland weed control because of its very low ecotoxicity; imazapyr is as low or lower in wildlife toxicity, but its activity is more persistent (it is not immobilized by fine sediment, and can spread in surface water or groundwater while active). Other herbicides may be appropriate for specific weed species (see section 5.1).

Herbicides may be applied by hand-sprayer, backpack sprayer, by pressurized wicks (wiping), or direct cut-stump application of herbicide solution. Cut-stump applications, or application of spray to short, dense regrowth of cut or mown weeds, generally minimizes the total amount of herbicide needed to treat a patch, and minimizes risk of non-target impacts. Timing of herbicide application is a balance between maximizing receptivity of target plants (physiological activity, green leaf area), and minimizing receptivity of non-target plants to overspray or drift (treating targets after leaf senescence of non-target adjacent plants, or covering them with tarps). Because of the sensitivity of local wetlands and native plant communities, herbicide application should be guided by knowledgeable botanists closely familiar with local vegetation, to avoid non-target impacts.

4.2.6. Experimental soil conditioning (nitrogen immobilization/carbon addition, acidification)

Soils that have become nutrient-enriched because of past agricultural practices (residual fertilizer effects, manuring, cattle ranching) or nitrogen-fixing plant invasions can take many decades to return to pre-perturbation soil nitrogen levels. Non-native weedy vegetation is often competitively superior at high nutrient levels, but less so at low soil nutrient levels. Because many of the Pitkin Marsh and Cunningham Marsh species of concern are typically
associated with nutrient-poor, unproductive, acidic wetlands, there is justification for experimenting with productivity of vegetation by modifying soil acidity, soil nutrient availability (nutrient immobilization, or “reverse-fertilizer”). These types of techniques have been used with some efficacy in restoration of analogous acidophilic terrestrial communities such as heath (Isselstein et al. 2002, Owen et al. 1999, 2001, Marrs 1995).

Soil acidification is used to restore naturally acidic, often nutrient-poor soils that have been ameliorated and enriched by agricultural land use history. Soil acidification amendments add organic or mineral acids in bulk, altering the availability of some nutrients in the soil. Restoration of acidic soil reaction liberates some plant nutrients, and immobilizes others (transforming them to insoluble forms), and makes some, like calcium, more susceptible to net loss by leaching. It is likely that existing surface soils remain acidic, but it is possible that extreme acidity (natural for Sebastopol and Goldridge sandy loams) may have become buffered by past agricultural practices, or compensated by agriculturally enhanced soil fertility (cattle manure, runoff, enriched groundwater, alluvial sedimentation in eroded watersheds, past liming, etc.). Surface soil and subsoil sampling for pH (acidity/alkalinity) and available total nitrogen should be performed in terrestrial, wetland, and ecotonal habitats of the site. Soil testing can be coordinated through county agricultural extension services or private soil consulting companies.

Soil acidifying agents include (a) sulfur compounds such as aluminum sulfate, iron sulfate; (b) acidifying organic matter such as bracken litter or sphagnum peat moss; (c) mixtures of organic materials and sulfates. The advantage of metal sulfates is that they are soluble and can be applied without soil disturbance. Organic materials require application either as a mulch, or require tillage to incorporate them in the soil profile, which is infeasible for intact perennial or woody wetland plant communities. County agricultural extension services also provide technical support to landowners seeking guidance for application rates of acidifying amendments needed to achieve target pH (although to buffer extreme pH for agricultural fertility; for local wetland restoration, the opposite is the objective).

Nitrogen immobilization by altering soil carbon:nitrogen (C:N) ratios has been used to reduce the competitive advantage of some invasive perennial grasses in wetland sedge meadows. Adding slow-decomposing high-carbon organic materials such as fine sawdust (or milled rice hulls) to soil mixtures in greenhouse conditions immobilizes nitrogen (“anti-fertilizer” effect), and favors competition of stress-tolerant, slower-growing sedges in mixtures with fast-growing invasive grasses (Perry and Galatowitch 2002). Sawdust can be incorporated in grassland soils by tillage, but tillage is probably inadvisable for mixed native/nonnative grasslands with significant populations of perennial bulbs, forbs, and graminoids. Also, reducing bulk density of soils by tillage favors root penetration by fast-growing annual grasses at the same time it uproots perennial grass root systems. Tillage may therefore offset some of the potential benefits of carbon addition, and is clearly not feasible for intact grassland, shrub, or wetland vegetation.
Adding sawdust to intact soils and vegetation as a mulch may have different effects than mixing it in the soil profile by tillage. The high C:N material would be concentrated in surface layers as a thin mulch that would have intensive effects on the seedling layer, but slower or reduced impact on subsurface roots. As sawdust decomposes, soil nitrogen would become again available gradually, unless it is lost by volatilization during burning (section 6.4). The most practical experiments for carbon addition/nitrogen immobilization soil amendments may be (a) for use in disturbance patches (Sections 3.2.4., 3.2.5.), or (b) localized application in grasslands by raking. If significant benefits for native species recruitment (or suppression of non-native annuals) are indicated within 2 to 3 years after experimental sawdust application in patches, broadcast application of sawdust with agricultural equipment (for seeding or fertilizer application) may be feasible.

4.2.7. Leaf litter reduction or removal in patches

Leaf litter has important effects on plant growth, seedling recruitment, soil stabilization, and nutrient cycling. Removal of leaf litter in grasslands is associated with improved seedling recruitment and growth of native perennial bunchgrasses and forbs. Removal of leaf litter in riparian wetlands naturally occurs during extreme flood events that variously redistribute loose surface litter, expose surface soils, erode to subsoil, or deposit thick layers of sediment and organic debris. Artificial drainage and ditching probably has altered flood peaks of the Blucher Creek tributary on site, and fires are generally suppressed, so natural modes of occasional litter reduction have been damped or eliminated. The loss of litter-reducing processes may be partly compensated artificially through various grassland management techniques such as haying, burning (Section 6.4), or even manually raking patches of grassland after mowing. Litter removal in the riparian zone can also be achieved in patches by cutting vegetation (weed whips, mechanical weed-whackers) and removing litter with steel rakes in fall.

Litter removal patches should be monitored (or at least inspected) for at least one or two growing seasons to detect significant increases in either native or non-native plants. If non-native invasive species predominate in litter removal patches, control of seed source populations should be performed before preparing further patches. If seed banks supply most invasive species recruitment (i.e., no local seed parent populations), experimental litter removal patches should be located elsewhere, or discontinued. Litter can be used as an inhibitory mulch over patches of treated non-native invasive plants (herbicide, cut, etc.), or may be disposed in deep shade of willow thickets.

4.2.8. Girdling or felling of non-native trees

Girdling is a simple, slow technique for killing large trees in place without felling, creating snags (standing dead trees; Figure 4). Girdling is accomplished by removing a strip of bark
and “underbark” (cambium layer, exposing true wood) around the tree, so that carbohydrate transfer from shoot to root is severed, and root systems die. Girdling is most effective on non-sprouting trees (ones that do not regenerate from cut or burned stumps by forming adventitious buds, or activating dormant buds in bark). Non-sprouting trees include pines and firs. Sprouting trees regenerate from proliferation of shoots below the girdle. Killing stump-sprouting trees requires one or more herbicide treatments of the tissues at the base of the freshly cut girdle. Girdling tools may include axes or chainsaws. The cambium layer is most readily detached from true wood when sap is flowing in abundance (high soil moisture) and cambium is actively growing in spring to early summer. Girdling is an alternative to tree removal when disposal or transportation of felled trees is too disturbing to vegetation, too expensive, or impractical. It is appropriate mainly to low-hazard settings with limited or no public access. Snags created by girdling often develop high wildlife value as they decay. Softwoods like pines rapidly decay into valuable wildlife habitat as snags.

4.2.9. Transplantation and seeding.

Many native plant communities degraded by past land use practices, or subjected to long-term dominance by invasive non-native species, may become native seed-limited, especially for slow-dispersing widespread species, or rare species with no proximate or local populations. Many seed introduction or seed bank augmentation (seed addition) studies suggest that in some communities, recovery of plant populations are in fact seed-limited (Turnbull et al. 2000). Recent studies in California serpentine grasslands have supported this (Seabloom et al. 2003). Restoration strategies for plant communities where seedling establishment is erratic (or dependent on infrequent climate events like late rains) may rely substantially on vegetative propagation of perennial forb, graminoid, or woody species, and transplanting limited numbers of field-grown or nursery-propagated stock to restoration sites. Both reintroduction methods are relevant to management of the CDFG easement site.

4.2.10. Seed bank probing, “prospecting”

Many apparently extirpated plant populations persist as dormant seed banks. Some rare plants or endangered species presumed to be extinct in the wild, such as Ventura Marsh milkvetch and Humboldt milkvetch (Astragalus pycnostachyus var. lanossimus, A. agnicidus), have been rediscovered re-emerging from dormant seed banks after decades, following disturbances to soil, light exposure, or both. In nearby Sonoma County sites, endangered species populations presumed extirpated have re-emerged after changes in land uses (e.g. regeneration of Limnanthes vinculans colonies following removal of hogs at a Cotati site circa 1992). For these reasons, it may be prudent to probe soil seed banks at sites where rare plants have been reported as recently (10-20 years) extirpated, such as the CDFG easement site. Methods for studying seed banks are too labor-intensive for broadcast searches of buried seed. Low-level patches of disturbance (slicing off vegetation at soil surface level to expose bare gaps; heavy raking) scattered over an area of potential or suspected occurrence is one practical approach to “prospecting” for missing species, providing disturbances do not interfere with
higher-priority management actions for extant populations of plants being conserved.

4.2.11. Establishment of buffers (weed seed dispersal, nutrient or sediment capture)

Buffer vegetation refers to establishment of any vegetation that mitigates or reduces the impact of an incompatible adjacent land use, process, nuisance, or activity. At the CDFG easement site, the primary nuisance that could be addressed by a buffer vegetation is nutrient-rich runoff and weed seed dispersal from intensive cattle ranching (Section 6.6) bordering a portion of the CDFG easement site. Planting a dense buffer strip to trap wind-blown seeds, capture and infiltrate runoff in litter and surface roots, and assimilate nutrients in fast-growing woody tissues, may mitigate adverse cattle influences to an unknown degree. Buffers of native shrubs may be installed along other parcel boundaries where changed land uses threaten to increase seed rain of weeds, or runoff of fertilizer or manure, into the conservation easement site.

4.2.12. Grassland management techniques

Grassland management techniques include mowing, haying, burning, and seeding/transplanting. Mowing timed to key stages of flowering and seed production can be used to reduce seed production of annual grasses over years. Haying is used to remove accumulated leaf litter and reduce nutrient availability (soil impoverishment) that tends to favor competition of non-native annual grasses and forbs. Burning has similar effects on nutrient pools, above-ground biomass and litter, and if properly managed, can stimulate growth and structure of native perennial grasses and forb communities. Burning can sometimes inhibit some non-native perennial grasses that are less tolerant than native grasses. Seeding and transplanting species-impoverished grasslands (following decades of dominance by non-native grasses) with native species can also assist diversification of restored grasslands. These techniques are discussed in detail in Section 6.4.1.

4.3. Practical inspection and monitoring for annual management

Long-term management of the site should include continuity of careful observation, and long-term inspection, or quantitative monitoring to the extent feasible. Documentation of field trials and practical experiments that apply combinations of vegetation management techniques, and their results, is necessary for assessment and adaptation of management methods over generations of stewardship. Levels of volunteer monitoring efforts, or graduate student research efforts, tend to occur as relatively short-term peaks of intensive data collection and analysis, but with low continuity over periods of time beyond 5 years. Similarly, adequate funding for intensive monitoring may not be able to be sustained over long time periods because of cyclic trends in funding and emphasis on data collection and analysis. Because the most significant results of restoration and management actions manifest over periods of time on the order of 5 to 20 years or more, long-term continuity of observations and documentation is more important than short-term, intensive data collection.
and analysis. It is important to design a minimum base level of inspection, reporting, or monitoring that can be sustained over periods of time longer than 5 years. Data collection techniques and sampling methods must be simple and robust enough to be stable over long-term changes in scientific methodological standards.

Basic records of vegetation management activities should involve accurate, recoverable map locations, descriptions of management actions, and photographic documentation of immediate pre- and post-implementation conditions. Basic descriptive or quantitative information should be provided in all individual project records: area treated, dates of treatment, semi-quantitative or quantitative description of vegetation, quantities of materials added or removed, changes in dimensions of features modified, numbers of individuals treated.

Quantitative monitoring of baseline condition change for a 19-acre site with multiple plant community types may be impractical, given the potential for competition for funding and labor between monitoring and practical management. Qualitative or semi-quantitative monitoring with limited sampling of baseline vegetation conditions may be achieved by conducting permanent plot relevés (Bonham 1989) at fixed dates or phenological stages of indicator species. Releve data should include accurate species lists (with vouchers, if possible), and ranked cover/abundance classes for all species recorded. For woody or clonal species, ranked colony size-classes or height-classes may be recorded efficiently as well.

Demographic monitoring is generally very labor-intensive, and should be considered only for highest priority species and analyses. Few demographic studies can be sustained in the long-term as populations and data management needs grow over time. Basic, coarse demographic data such as size-classes of reproductive individuals, number of fruits (whole population or subsampled areas), subsampled numbers of seeds per fruit, and number of colonies (ranked size-class), are among basic demographic variables.

Establishment of a permanent large-scale grid system, tied to permanent stakes or fenceline or site boundary markers, would facilitate recovery of plot location or plant population information in the long-term. A grid system would also facilitate implementation and documentation of vegetation management activities. If GPS data and technical support are available within budget constraints for management, documentation of key reference points or population boundaries would benefit long-term monitoring.

Collaboration with local public school, college, or university science programs can provide valuable volunteer monitoring labor sources, but only if data quality can be provided adequate supervision. The quality (and thus long-term value) of citizen-based, non-expert data collection can vary widely. The most successful programs are likely to emerge under the supervision of experts from local universities or resource agencies with long-term commitment to the site.
5.0 Species-specific management practices

5.1. Non-native nuisance or invasive species

5.1.1. *Centaurea solstitialis, C. melitensis* (star-thistles)

Starthistles (*Centaurea solstitialis, C. melitensis*) are present at the CDFG easement site, and are generally present in the site vicinity. It is likely that starthistles are also present at the intensively trampled adjacent cattle grazing site. Other adjacent residential sites appear to be regularly mowed, and are unlikely to be long-term seed sources. Starthistles are noxious, highly invasive forbs in grasslands of southern Sonoma County and most of California’s grasslands. In the absence of factors that suppress their spread (frequent fire, sheep grazing, haying or mowing in early summer), they can rapidly expand from minor populations to major infestations (DiTomaso and Gerlach 2000). Starthistles should be a high priority for control even if local population sizes are (temporarily) small.

Small populations of starthistles can be controlled or eradicated by eliminating reproductive plants very soon after they begin to flower in early summer. Starthistles have relatively short-lived seed banks, so preventing seed production for several successive years can effectively control local populations with low rates if immigration from adjacent populations (DiTomaso and Gerlach 2000, DiTomaso 2003, DiTomaso et al. 1999)

Adjacent land uses probably constrain most of the maintenance methods for controlling starthistle, such as periodic controlled burns or seasonal sheep grazing. The conservation purposes of the easement site are incompatible with methods such as tillage or extensive use of herbicides. Mowing or haying (mowing plus harvest of cut hay; see section 6.4.1), and local weed cutting are potentially effective control techniques that can prevent seed set of starthistle if they are closely timed to the early stages of flowering. Starthistles can begin to set viable seed (germinable after cutting seedheads) within 2 weeks of the onset of flowering (DiTomaso and Gerlach 2000, DiTomaso et al. 1999). If cutting occurs during early flowering, and main shoots are severed close to the ground (below stem leaves), basal branching and re-flowering can be minimized. Repeated cutting during the summer may be necessary if basal shoots develop after initial cutting. Basal shoots tend to develop decumbent to ascending branches low to the ground, making subsequent mowing less effective.

If starthistles are cut or mowed after many weeks of flowering, cut shoots will retain seedheads containing viable seed. Mid-summer cutting must be combined with thorough removal of cut stems, or mowing/cutting will not significantly reduce population size the following year. Premature cutting in April/May (the usual time for mowing non-native annual grasses; see section 6.4.1) is unlikely to control starthistles; instead, it encourages basal branching and a lower-growing habit. Separate late spring/early summer mowing or other treatments are needed to combat starthistle.
Long-term cultural control of starthistle may include reduction of annual grass dominance, and increased extent of rhizomatous, sod-forming native perennial graminoid species such as *Leymus triticoides* and *Carex barbarae* (both reported from the site) and perennial bunchgrasses (see section 6.4.1.). Long-term cultural control may also include increasing local barriers or impediments to dispersal from adjacent parcel populations, if any occur. Planting native shrub borders to adjacent cattle-grazed land would be a useful long-term partial barrier to dispersal (see section 6.6).

5.1.2. *Cirsium vulgare* (bull thistle)

Bull thistle is present in the soft rush/sedge marsh below the concrete culvert, and in the ruderal grassland between the marsh and the fenced cattle grazing area on an adjacent parcel. It appears likely that the highly disturbed, weedy, nutrient-enriched cattle grazing area provides a persistent local seed source for the marsh, particularly along the edges where cattle cannot trample thistles. Bull thistle appears to be uncommon in other parts of the CDFG easement site. Bull thistle readily establishes in seasonal wetlands, particularly where small disturbance patches occur. Deer paths across the marsh to the white poplar thicket are likely to provide trampling disturbances for bull thistle establishment. While it is not currently abundant, it has potential to proliferate rapidly in either natural or artificial disturbances in the marsh, to the detriment of native species that would otherwise be able to colonize disturbance gaps. It would be desirable to allow for the possibility of natural gap dynamics in the marsh to occur in the future with native species, so control of bull thistle may be warranted.

Bull thistle in the marsh itself is likely to remain either in the seedling stages (or unemerged) during the spring in wet years (late rains), and grow rapidly as the marsh drains in early summer. In dry years, development of rosettes is likely to occur earlier. Bull thistle in the marsh can be controlled during large rosette or bolting (stem elongation) stages before flowering by uprooting the taproot with a narrow tile spade. Severing the taproot near the surface, or crushing the central apex will risk regeneration and delayed flowering. Seed sources in the adjacent terrestrial grassland may be controlled during bolting stages with a weed whip, mattock, or tile spade. Cooperation with adjacent landowners to allow mowing or mechanical weed cutting of flowering bull thistle would assist control. Planting tall native shrub borders to adjacent cattle-grazed land would be a useful long-term partial barrier to dispersal (see section 6.6).

5.1.3. *Eucalyptus globulus* (blue gum)

There are several large, old, blue gum trees near the riparian/upland transition zone at the upstream end of the CDFG easement site. The old trees are probably portions of historic windbreak plantings. There are also numerous, scattered immature blue gums in and around
the willow riparian thicket immediately around the old trees. There is little current evidence of frequent seedling establishment in recent years. Within a few decades, however, these fast-growing trees (benefiting from riparian soil moisture) may form a dense closed-canopy grove, and increase seed rain on remaining portions of the riparian corridor. The existing old blue gums have a semi-open canopy, and are spaced so that no intense shade or litter impacts are evident in the vegetation below them. There is a moderate amount of tree litter (leaf, branch, limb), but it appears to be little or no impediment for the growth of native hawthorn and other, more common native plants. The mature gums may be used as raptor perches (e.g. red-tailed hawks and other buteos).

Because of the high cost and potential disturbance of removing large, old blue gums, and the relatively small number and low density of mature trees, I recommend that control efforts instead be focused on removal of the smaller, fast-growing young trees (juvenile or transitional juvenile/mature growth phase). Removal of immature blue gums would also be less likely to raise concerns or objections from adjacent landowners. Immature blue gums can be felled, bucked (sectioned), and hauled away as firewood. Eucalyptus wood is very slow to decompose, and so felled trunks should not be left on site. Thick blue gum slash (fine branches with persistent attached foliage) can be disposed on top of Himalayan blackberry stands, or along their edges, to smother or inhibit growth. Blue gum stumps must be treated with herbicide when sucker shoots proliferate, or directly on freshly cut stumps (especially near bark). Waxy leaves of blue gum may repel wetting by herbicide solutions, so some mechanical scarification of sucker foliage may be useful to ensure uptake of herbicide.

5.1.4. *Pinus radiata* (Monterey pine)

*Pinus radiata* (Monterey pine; reported as *P. muricata* in some past CNPS lists) occurs on the site mature trees remnant from old agricultural windbreak plantings located just above the riparian willow thickets and ecotone. Trees are mostly mature, senescent, or dead (snags). They are not strongly invasive (few juveniles), but the space they occupy would be better allocated to native oak woodland over many decades. Because felling and removal may have excessive short-term costs or impacts, and because snags (wildlife trees for owls, woodpeckers) are scarce and valuable, accelerating death of mature or senescent trees is recommended. Girdling methods (Section 4.2.8), stand-level management (Section 6.5) and oak management (Section 6.4) are basic components of the Monterey pine management strategy.

5.1.5. *Populus alba* (white poplar)

European white poplar forms a clonal thicket in the riparian zone below the concrete culvert and rush-sedge marsh on the CDFG easement site. White poplar requires summer soil moisture, and so is seldom an invasive species in the Bay Area or coastal California. It is likely to remain confined to the riparian zone, but its local dominance and dense shade and
cover is detrimental to plant community conservation objectives. In addition, the dense white poplar grove appears to be preferred cover for deer: a concentration of heavily trampled deer trails that form a network centered on the poplar grove. The proximity of preferred deer cover to sensitive lily populations may expose lilies to elevated risk of browsing injury during flowering or seed set. The space occupied by the poplar grove would readily be vegetated by willows, which would have higher wildlife habitat value.

Poplars cannot be killed by felling: they readily sucker and stump-sprout, so cutting would result in a coppiced thicket of greater density. Poplars could be killed by frilling/herbicide or cut-stump/herbicide application in the summer, when trees are actively growing, and there is no flowing or standing water. Aquatic formulations of glyphosate (lacking surfactant) may be used within the wetland area. Disposal of poplar slash should be located directly on the nearest Himalayan blackberry thickets.

5.1.6. *Rubus discolor* (Himalayan blackberry)

Himalayan blackberry is one of the dominant plants of the riparian zone and its edges. It also occurs in separate thickets around the bases of dead pines above the wetland area. In semi-shaded riparian wetlands, it reaches heights well above 2 m, and smothers nearly all other vegetation below its dense, closed canopy.

Himalayan blackberry is extremely tenacious, readily regenerating from cut stumps and “suckering” from adventitious shoot buds on root segments. Manual removal is feasible, but very difficult because of its flexible, thorny canes, and recalcitrant below-ground suckers from severed roots or roostocks (below-ground “trunks”). Machetes, hedge shears, and lopping shears can be used to cut back thickets to basal roostocks. Basal rootstocks can be left intact to proliferate suckers accessible for subsequent treatment with herbicide (see below), or they can be cut back by wrenches or mattocks. Regeneration by suckers on remaining below-ground severed roots or roostocks is almost inevitable even with thorough efforts.

Mechanical removal of Himalayan blackberry is feasible for isolated terrestrial stands, but would be undesirable for removal of extensive thickets below willow canopies, where access is limited, and disturbances need to be limited in duration and size. Herbicide use within aquatic riparian habitats in California must be restricted to aquatic formulations of glyphosate, and only herbicides with minimal toxicity to fish and wildlife (like glyphosate) would be compatible with wildlife management in the vicinity of riparian corridors. Other herbicides, such as triclopyr (Garlon®) may be used effectively on blackberry in terrestrial vegetation.

Glyphosate is only moderately effective on the complex branching structure of intact Himalayan blackberry stands (pers. observ.). Partial coverage of stands composed of physiologically independent units (separate root systems) often appears to allow partial
survival of treated stands. In addition, the large, complex branching structures of Himalayan blackberry makes both spray or wick application of glyphosate difficult. Glyphosate can be highly effective at killing rootstocks during early stages of regeneration by suckering after mechanical removal. Combining mechanical removal with (post-suckering) glyphosate application minimizes the hazards of overspray (non-target herbicide damage) in riparian habitats, and minimizes the total dose of herbicide needed to achieve high mortality. Similarly, glyphosate may be used as a follow-up treatment for mechanical removal.

Intact Himalayan blackberry remains green and physiologically receptive to glyphosate even in late summer and early fall, when leaves of most native plants (and all sensitive species) are in relatively advanced stages of senescence. Himalayan blackberry cut back to stumps in mid-summer would support rapid growth of physiologically active, green, “soft” sucker shoots in late summer and early fall. Combining mid-summer manual removal of Himalayan blackberry in riparian thickets, followed by late-summer/early fall spot-spraying of resprouts with glyphosate and low-toxicity surfactants (like Agridex®) would minimize non-target herbicide impacts. Glyphosate is inactivated by contact with soil, and would therefore have no significant residual herbicide effects downstream, or persistent local effects on soil, if it is applied at label concentrations and dosages.

Mechanical removal of upland or riparian/upland transition Himalayan blackberry stands may be combined with experimental vegetation treatments that involve exposure of low-nutrient, low-organic subsoils (see Section 4.2.6.).

5.1.7. European grasses

European annual grasses (Avena, Briza, Bromus, spp.) and perennial grasses (Agrostis, Anthoxanthum, Dactylis, Holcus, Lolium spp.) are proposed for general management by mowing and haying regimes prescribed for native grassland restoration (Section 4.2.12). Sustained mowing/haying regimes properly timed, and implemented with sufficient frequency, stand a good chance of significantly reducing non-native annual grasses and promoting native perennial grasses, combined with supplemental restoration techniques explained in Section 6.4. Perennial non-native grasses may not be as readily controlled by mowing/haying as annual species. Mesic-preference non-native perennial grasses such as Holcus, Dactylis, are likely to occur not just in grasslands, but in mesic riparian transition zones as well. There they may require spot-treatment with herbicides or manual removal if their spread is progressive, and densities are problematic. Perennial grasses such as terrestrial forms of Agrostis stolonifera, or Anthoxanthum, may behave ecologically much like native perennial bunchgrasses, and may be effectively permanent naturalized components of the grassland, since their phenology and growth-form and environmental tolerances overlap considerably with native perennial grasses. To some extent, they may be competitively displaced by rhizomatous sod-forming native perennial grasses in mesic portions of the grassland (see Section 6.4.).
5.2. Native species with special conservation status

The following native plant species have been suggested by CNPS as candidates or priorities for conservation at the CDFG easement site. Each is evaluated for site-specific appropriate conservation actions, and feasible population management activities are prescribed where they are justified.

5.2.1. *Alopecurus aequalis* var. *sonomensis* (Sonoma alopecurus, Sonoma foxtail)

*Alopecurus aequalis* Sobol. var. *sonomensis* Rubtzoff (Sonoma alopecurus) is currently treated as synonymous with the variable, wide-ranging species in the Jepson Manual (Crins 1993), but is recognized as a distinctive geographic population if not a formal taxonomically distinct rank. It was originally distinguished by its long awns, more robust and erect habit, wider blades, and the isolation of its Marin and Sonoma populations from the main range of the common species. CNPS and the U.S. Fish and Wildlife Service still recognize the Marin-Sonoma populations as a distinct variety listed as endangered. In southern Sonoma County, historic populations of the species that may be referred to this variety were collected from, Pitkin Marsh, Cunningham Marsh, Guerneville (type locality) and possibly other sites as well (Best et al. 1996).

Sonoma alopecurus is a short-lived perennial cespitose (tufted) grass. It has been reported from Cunningham Marsh north of Blucher Creek by Howell in 1947. It may be extirpated, or persisting as either a seed bank or undetected or intermittent population, on or off the easement site. Sonoma alopecurus is relatively well-studied at Point Reyes National Seashore, where multiple localities have been surveyed or subjected to experimental manipulations (Gennett et al. 2004). At Point Reyes (the extant core population), it typically occurs in seasonal wetlands of sandy grasslands, emergent beds of muddy or sandy seasonal shallow ponds, or active (disturbed) alluvial fans and deltas of small seasonal streams that discharge into isolated basins or ponds (Gennett et al. 2004, Baye pers. observ.). Sonoma alopecurus can produce abundant viable seed when grown in conditions with ample moisture and low competition (Gennett et al. 2004). It is unlikely to persist in dense, stable moist grassland with abundant competition. There is no evidence to suggest that Sonoma alopecurus is ecologically or reproductively distinct from *Alopecurus aequalis*, a widespread, common plant of wet meadows and similar wetland habitats.

Sonoma alopecurus can be cultivated to produce abundant seed, and it has been successfully seeded into experimentally manipulated natural vegetation at Point Reyes. Despite the high feasibility for reintroduction to presumed extirpated historic sites, it may be counterproductive to do so. Because each historic population in southern Sonoma Co was probably long isolated from others, it is reasonable to presume that significant genetic differentiation occurred among populations. If non-local seed sources are used for
reintroduction, it would likely corrupt any revived populations from dormant seed banks, and could obscure distort future scientific investigations if documentation of reintroduction is lost or underreported. For these reasons, conservation priorities for the CDFG easement sites should be (1) annual searches for rediscovered or re-emergent populations, especially near moist soil disturbances; and (2) receptivity to establishment of populations discovered within other portions of Cunningham Marsh. Reintroduction of offsite populations should be focused on new, suitable localities relatively close to their sources. Establishment of new populations should experiment with (1) variable local disturbance regimes or hydrologic regimes that regenerate gaps in sandy or muddy substrate (sediment deposition or bare areas); (2) production of abundant seed from diverse maternal parents in cultivation, for use in seeding on site; and (3) very high initial seeding rates on site.

5.2.2. *Carex* species (sedges)

Numerous *Carex* sedge species have been reported from Cunningham Marsh, and many specifically the CDFG easement site. Obligate wetland sedges (at least seasonally saturated soils) include the rhizomatous slough sedge (*Carex obnupta*, common and widespread in California) and Nebraska sedge (*C. nebraskensis*, regionally uncommon; reportedly sterile plants reported by J.T. Howell; Best et al. 1996)), the tufted *C. gymnodynama* (as “*Carex gymnodynama*” in the CNPS list; regionally rare), *C. dudleya* (regionally common) *C. ovalis* (regionally uncommon), and *C. densa* (regionally common). Santa Barbara sedge (*C. barbarae*) is a rhizomatous, colony-forming sedge found in seasonally wet to well-drained mesic hillslopes or alluvial grasslands. *Carex albida*, federally endangered and endemic to Pitkin Marsh, is a tufted, non-rhizomatous sedge found in sphagnum peat.

Slough sedge (*C. obnupta*) is relatively common in freshwater marsh at the CDFG easement site, and does not appear to be dispersal-limited in the very wet soils (seeps, marshes) it depends on. There are few strongly invasive plants in the wetland zones in which it is most competitive, so it would have limited value for propagation or transplanting. Santa Barbara sedge (*C. barbarae*) has a potentially wider range on the site, and may be useful and appropriate for revegetating mesic grassland or grassland/riparian ecotone areas (seasonally saturated soils) where patches of non-native vegetation have been removed, particularly Himalayan blackberry. Suitable planting sites for vegetative divisions (rooted clusters of shoots, and rhizome segments with tillers or terminal leafy shoots, foliage partly pruned) would include any semi-shaded or exposed soils that are wet or moist in March and April, and still moist in May. Transplanting of divisions should be restricted to cold, wet weather in late November-January to allow rooting before significant leaf elongation occurs. Santa Barbara sedge transplants may also be attempted in relatively dry upland grasslands. Rhizomatous sedges should not generally be planted among colonies of Pitkin Marsh lily, since dense sods of colonial sedges may interfere with growth of lily clones. Multiple clones should be interplanted or planted adjacent to reduce pollen limitation due to self-incompatibility.
Cespitose wetland sedges species may also be spread by divisions in winter. If their source populations are small, they should be propagated offsite as labeled, distinct clonal lineages, and replanted on site as small colonies of mixed clones (again, to avoid pollen limitation of self-incompatible clones).

It is unlikely that the CDFG easement site would support the sphagnum peat habitat of Carex albida at Pitkin Marsh, but the species was formerly found at other localities that probably supported it on other substrate types. If the survival of the C. albida population of Pitkin Marsh remains insecure, it should be propagated (with landowner permission) from a sample of at least 25 to 50 seed-parents and clones from the entire population, and propagated by seed and divisions in controlled conditions. An experimental population could be transplanted to a modified portion of the rush-sedge marsh (or site of white poplar removal adjacent to it), where overgrowth of willow and rushes are managed. Treating the C. albida transplant sites with soil amendments that lower pH and immobilize soil nitrogen (Section 4.2.3) after transplants are established for one growing season, with clipping to control competition initially, is recommended. Introduction of C. albida from Pitkin Marsh should be viewed as experimental because of uncertainty of long-term stability of the modified receptor habitat and introduced population; it is not a surrogate for sustaining the source population. Because of the endangered status of this species, a full reintroduction and monitoring plan would probably be required by CDFG. Intrastate translocation of federally endangered plants from non-federal jurisdiction, conducted without criminal trespass or violation of state laws, is not prohibited or regulated by the federal Endangered Species Act.

5.2.3. Calamagrostis bolanderi (Bolander's reedgrass)

Bolander’s reed-grass (Calamagrostis bolanderi), is a rare perennial rhizomatous, colony-forming grass of bogs and wet meadows. It is morphologically and ecologically similar to the common C. nutkaensis. I did not locate any plants during May and September 2004 visits, but they have reportedly been identified tentatively on the site in recently years. If the local C. bolanderi population not extirpated, but is reduced to a single clone, it is likely that it would be circumstantially seed-sterile, owing to self-incompatibility. If multiple clones occur, it would be more feasible to restore a reproductive population. Natural seedling recruitment should be expected to be infrequent, however. Calamagrostis species are readily propagated by either rhizome divisions or divisions of rooted shoot clusters, during winter dormancy. In cultivation for propagation of clones, they grow vigorously in response to liberal addition of soil nitrogen, but their success in competition in natural conditions is associated with nutrient-poor, acidic, wet sandy or organic soils.

The CDFG easement site should be searched for putative Calamagrostis. If diagnostic panicles are lacking, and there is no evidence of recent past seed-heads, the plant(s) should be propagated and grown offsite in cultivation to hasten attainment of flowering-size clones.
Calamagrostis can be readily grown in large tubs alternately saturated and drained but moist. If clones are verified as Calamagrostis, they should be divided and repatriated to the site. Site selection for transplants should include a range of moisture conditions, including the open sedge/rush marsh below the culvert, and the wet/mesic rush/blackberry vegetation under dappled tree shade. Transplants should be planted into disturbed gaps about 20 cm wide, and mulched with local duff. Location of transplants should be flagged, recorded, and monitored for survivorship.

No unrelated coastal populations of C. bolanderi from northwestern Sonoma Co (the main location) or west Marin County should be introduced to the CDFG easement site at Cunningham Marsh. The conservation value of protecting an ancient, disjunct, isolated population would be corrupted by introducing genotypes from core coastal populations.

5.2.4. Campanula californica (California bellflower)

California bellflower (Campanula californica) is a slender-stemmed decumbent perennial herb of wet meadows, woodlands, and wet forest gaps along the Marin, Sonoma, and southern Mendocino coast. Some CNPS members have nominated it as a candidate species for introduction to the CDFG easement site. I have found no historic records of this species from Cunningham Marsh, but it has been reported from Pitkin Marsh (Best et al. 1996).

I recommend a very conservative approach to new introduction of C. californica to the CDFG easement site. The strongest justification for introducing it there would be if it were rediscovered at Pitkin Marsh or Perry Marsh and found to be in severe decline (marginal viability), and infeasible to recover at those locations. In that case, if better potential sustainable habitat were available at the CDFG easement site, it might warrant introduction. Otherwise, translocation of C. californica from other localities and vegetation types in this species’ range (almost all near the coast) to the CDFG easement site should be prohibited. Introducing a population from outside the Pitkin/Perry/Cunningham complex should not be considered to be a conservation measure for the species. Even introducing a population from Perry or Pitkin Marsh implies a corruption of biogeographic scientific data for future research if documentation of the introduction is lost or obscured over time.

If Campanula californica is introduced to the site from a local donor marsh, it can be propagated in early spring by division of naturally rooted prostrate stems in moist duff, somewhat like the more common woodland species C. prenanthoides. If seed are found, they may also be treated like C. prenanthoides (sown on stratified milled sphagnum/seedling compost in semi-shaded outdoor beds in winter). Seedlings and young plants, however, would require constant soil moisture and high organic matter (milled sphagnum moss and compost) and partial shade.
5.2.5. *Castilleja uliginosa* (*C. miniata*) (Pitkin Marsh paintbrush)

Pitkin Marsh paintbrush (*C. uliginosa*) has been in cultivation at the U.C. Botanical Garden at Berkeley, originally under the direction of the late Robert Ornduff, and more recently maintained by Holly Forbes. The status of the disjunct Pitkin Marsh population in the wild is uncertain. There are no historic records of this entity at Cunningham Marsh, and it is doubtful that there is suitable habitat among the productive marsh and riparian woodland at the CDFG easement site. The taxonomy of the Pitkin Marsh *C. uliginosa* is also uncertain: it may be a variable disjunct population of *C. miniata* (Hickman 1993) or something closely related to it. Given the difficulty of cultivating or transplanting hemiparasitic Castillejas even in favorable circumstances, and the doubtful suitability of the endemic Pitkin Marsh population at Cunningham, attempting to conserve translocated populations at Cunningham marsh should be a low priority.

5.2.6. *Cornus sericea* (American dogwood)

*Cornus sericea* itself has no special conservation status, but it is likely to suffer declines locally because of competition with overproductive, monotypic willow thickets, and interference with seedling regeneration by Himalayan blackberry. It naturally occurs in riparian vegetation with intermittent disturbance by erosion. *C. sericea* can be propagated readily by hardwood cuttings in fall, or by seed. If disturbance patches created by removal of Himalayan blackberry are available within the mesic to wetland riparian zone, it would be appropriate to include propagated transplants of *C. sericea* in dormant-season planting plans.

5.2.7. *Isoetes nuttallii* (Nuttall’s quillwort)

*Isoetes nuttallii* is an uncommon but relatively widespread non-flowering plant found in undisturbed vernal marshes, pond edges, and deep vernal pools. It is seldom found in highly productive marshes with fertile or disturbed soils, and is expected to decline in competition with invasive perennial marsh plants in eutrophic wetlands like the Laguna de Santa Rosa. If excavated perennial to late-emergent seasonal wetlands are constructed at the CDFG easement site, *Isoetes nuttallii* should be considered as a species suitable for translocation to the site, if it is not still present. Suitable donor populations can probably be secured from the southern Santa Rosa plain.

5.2.8. *Lilium pardalinum* ssp. *pardalinum* (Pitkin Marsh lily)

Multiple clonal populations of *Lilium pardalinum* ssp. *pitkinense* (the rare endemic Pitkin Marsh lily) occur at the the CDFG easement site, and they were among the primary justifications for the acquisition (Figure 7). Both *L. pardalinum* ssp. *pardalinum* (leopard lily) and *L. p.* ssp. *pitkinense* have been reported at Pitkin Marsh (Best et al. 1996), so this site is the only one where long-term conservation of “pure” Pitkin Marsh lily is probably most
Past threats to the local Pitkin Marsh lily (lily) population included excessive cattle trampling and browsing, deer browsing, interference by invasive plants like Himalayan blackberry, and depletion of the population by horticultural overcollection of bulb-like rhizomes. It is uncertain whether hydrologic changes, such as changes of flood frequency, intensity, sedimentation, or groundwater depression, have reduced the viability of the population. Recent clonal expansion and recruitment of new clonal populations strongly suggest that site conditions are currently suitable for regeneration or expansion of the population at this site. Deer browsing and overgrowth by Himalayan blackberry appear to be the most significant threats. These have been mitigated by poultry-mesh exclosures and removal of non-native blackberry thickets near the lily colonies by volunteer CNPS site stewards.

Abundant production of capsules with high seed set (apparently viable, filled seeds) was evident in protected populations in September 2004 (P. Baye, pers. observ.). New seedling populations have been detected and protected by exclosures as they are discovered (B. Young, CNPS, pers. comm. 2004). No monitoring data have been available to date, but demographic data collection was initiated in 2004 by CNPS volunteers.

The proposed vegetation management strategy and actions aimed at conserving Pitkin Marsh lily at this site rely substantially on recovery strategies for a related and ecologically similar rare California lily, western lily (Lilium occidentale). Western lily also occurs in sandy acidic wetland soils in coastal scrub and forest gaps. Pitkin Marsh lily was formerly treated as a distinct species closely related to L. occidentale, differing in having “… short stolons between the rhizome sections” (Munz 1959). Western lily occurs in acid, sandy or peaty successional ericaceous scrub, coastal sedge meadows, and grazed pastures from coastal Humboldt County to Oregon (Guerrant et al. 1999). Peter Rubtzoff reported an isolated population of L. occidentale at the Ledum Swamp (ericaceous, acid fen-carr) at Point Reyes, Marin County (Rubtzoff 1953, Howell 1970), ecologically similar to Pitkin Marsh in some respects. The western lily recovery approach has been partially adapted to Cunningham Marsh for this vegetation management plan, but modified to account for local site conditions, and to incorporate this author’s observations of wild northwestern Sonoma County riparian, seep/spring populations of L. pardalinum ssp. pardalinum, and experience with propagation of L. p. ssp. pardalinum and L. maritimum (another ecologically comparable rare lily species found in sandy, acid, coastal wetland soils in scrub and forest gaps near the coasts of Sonoma and Mendocino Counties).

Lilium occidentale, L. pardalinum, and L. maritimum appear to establish seedlings in relatively open vegetation or vegetation gaps, but persist as shrubby vegetation and overstory canopies develop during succession. Unproductive, acidic, sandy or peaty wetland soils naturally retard the pace of plant community succession, and may contribute to the persistence of lily populations. When mature, these lily species can regenerate through shrub canopies when mature, and display flowers above shrub canopies. Shrubs may partially protect plants from total defoliation by browsers, and provide additional structural support
(protection against lodging of stems by windstorms). Dense, tall shrub or tree thickets, however, may cause excessive shade and restrict visibility and access to the primary pollinators, hummingbirds (Guerrant et al. 1999; P. Baye, pers. observ.). In riparian corridors and floodplains, alternation between vegetation gaps (seedling habitat) and wet meadow/scrub succession may correspond to flood events and post-flood recovery cycles. Infrequent major disturbances such as fires in maturing wetland shrub communities may have been important for pulses of seedling recruitment in prehistoric times.

Existing deer exclosures for original clonal populations should should be either maintained or upgraded (section 4.2.1.) for easier use and future maintenance or expansion with clonal spread. If primary colonies are protected, variable levels of protection may be given to new colonies after their clonal spread is over 1 meter.

Because the population is currently producing abundant seed, and because it is more feasible to cultivate long-lived clones than store seeds in the long term, it would be prudent to develop an offsite “clone bank” of seedling-grown genotypes as a hedge against risks of population decline, catastrophic population loss, or loss of genetic diversity in the natural clonal population. The existing population, though currently healthy and apparently expanding, is still subject to failed or limited regeneration following severe droughts, pathogens, or excessively fueled fires (note proximity of large blue gums to some colonies). Replicate clones from maternally balanced multiple seed lines may be cultivated in one or more botanical gardens, or in gardens maintained by CNPS stewards with faithful maintenance of labels to distinguish genotypes and parentage.

Population size in the field should be increased first by clonal expansion of existing plants, facilitated by local removal of suppressive Himalayan blackberry (Sections 4.2.4., 4.2.5., 5.1.6.), and increased canopy gaps in willows (Section 4.2.2.). The next priority would be to facilitate seedling establishment of new colonies, allowing spontaneous recruitment of seedlings, but facilitating seed dispersal in to disturbance patches. Seedling sites may be promoted by raking leaf litter and scarifying soil in either open rush-dominated areas to expose loam soils (Sections 4.2.4., 4.2.7), or under existing azaleas that may act as future “nurse plants” (section 4.2.3; Figure 8). Site selection should focus on the wetland ecotone between willow thickets and the rush-blackberry wet meadow/scrub. Because dispersal of lily seed by gravity is very limited (long-distance dispersal may rely on infrequent flood events), random samples of seed capsules from multiple clones should be collected in fall and seeds should be sown directly into gaps. Seedling emergence should be monitored the following winter and spring; L. pardalinum seeds germinate slowly over winter, and emerge conspicuously in spring. Emergent seedlings or seedling colonies should be protected by exclosures (section 4.2.1.).

If direct seeding experiments into gaps is unsuccessful in a range of precipitation conditions (wet and dry springs), cultivation of seedlings may be used as a source of transplant material, to bypass the high-risk seedling stage of size-dependent juvenile mortality. Natural seedling
recruitment in situ, following artificial seed sowing, is preferable over direct transplanting of cultivated seedlings of wild parents. Natural selection can operate at the seed germination and seedling recruitment stages with direct seeding, but natural selection is effectively eliminated or distorted by transplanting cultivated seedlings. Cultivation of seedlings and even random selection of transplants risks imposing long-lasting genetic “imprinting” of clonal populations established. If nursery production is used, genetic guidelines for rare plants should be followed (Brown and Briggs 1991, Guerrant 1996) to ensure that artificial selection and genetic drift are avoided.

If lilies appear to be suffering from excessive competition by non-native vegetation after initial removal efforts, soil amendments to reduce soil productivity (nutrient immobilization, acidification; section 4.2.6) may be applied to established clones of new lily seedling-derived colonies once they are growing vigorously and clonally spreading. If the soil amendments inhibit competing vegetation more than lilies, they may be repeated with other seedling-derived colonies. If they inhibit growth and reproduction of lilies with minimal effect on competing vegetation, they should be discontinued.

Other than removal of highly invasive Himalayan blackberries, and creation of willow canopy gaps where needed, management of lilies should avoid intensive “gardening” of established clonal populations of lilies the wild. The purpose of vegetation management is not to sustain maximized growth or reproduction of lilies, but to ensure that the population persists, and continues to reproduce and evolve over time. Indiscriminate removal of competing vegetation is counterproductive if this is the objective.

Translocation of lilies from Pitkin Marsh to the CDFG easement site should be prohibited. These populations have naturally been long isolated, and should be kept so. If ex situ conservation populations are necessary to preserve the Pitkin Marsh population, they should be located where there are no natural populations. If the Pitkin Marsh population is reduced to very few individuals, it may be necessary to introduce them with some individuals from Cunningham to avoid inbreeding depression or excessive loss of genetic variability.

General monitoring recommendations that apply in part to Pitkin Marsh lily conservation are given at section 4.3.

Further taxonomic and genetic study of the CDFG easement site population of Pitkin Marsh lily is warranted, including comparison with regional populations of L. p. ssp. pardalinum, the Marin disjunct population of L. occidentale (if still extant), and the type population at Pitkin Marsh (if available for study). Like the mutable taxonomic status of some other “endemic” Cunningham and Pitkin Marsh plants (Potentilla hickmanii, Castilleja uliginosa, Alopecurus aequalis var. sonomensis), it is possible that further study of lily systematics may re-open the status of its taxonomic rank. Some of the distinctions between the Pitkin Marsh lily and its more wide-ranging sibling subspecies, like “usually in small clones” (Best et al. 1996), “weakly clonal” (Skinner 1993) and overlapping perianth segment and anther lengths,
warrant more sampling of populations in a wider array of field conditions. The existing managed wild populations of Pitkin Marsh lily at Cunningham exhibit clonal spread of at least 1 meter, and possibly several meters, comparable with leopard lily in natural riparian settings of northwestern Sonoma County. Relict populations of western lily (*Lilium occidentale*), a similar rare lily closely related to the Pitkin Marsh population (Munz 1959) have been reported as far south as Point Reyes. No molecular genetic analysis of Pitkin Marsh lily is yet available to test its relationships with other lilies in the region. Given the genus’ tendency to form variable or backcrossed hybrid populations (Skinner 1993), this distinctive disjunct population warrants special biogeographic and legal protective status, regardless of its taxonomic rank.

5.2.9. *Limnanthes vinculans* (Sebastopol meadowfoam)

Sebastopol meadowfoam has not been detected on the site in recent years. There is currently insufficient information to determine whether suitable microhabitats with prolonged winter-spring inundation, and absence of tall perennial vegetation, exist on the site, but based on dry-season inspections, it currently (2004 observations) seems doubtful. If Sebastopol meadowfoam is rediscovered on the site, it may require excavation of seasonal wetlands depressions that remain flooded well into early spring, but do not develop tall emergent perennial marsh vegetation.

5.2.10. *Perideridia gairdneri* (Gairdner’s yampah)

If there are persistent populations of Gairdner’s yampah in the sites’ mesic grasslands, they may benefit from the grassland management regime prescribed in section 6.4.1., but they may be subject to short-term impacts during phases of intensive spring mowing and haying to reduce non-native annual grasses. If plants are detected, they may be staked and manually weeded to avoid mowing during grassland restoration. In contrast, yampah would likely benefit from dry-season controlled burns. Population size may be augmented by (a) seed collection and propagation, followed by (b) transplanting into mesic grasslands after achievement of basic perennial grassland restoration objectives.

5.2.11 *Quercus* species (oaks)

Refer to section 6.4.2. for management of oaks within grassland vegetation of the site.

5.2.12. *Rhododendron occidentale* (western azalea)

Western azalea may be propagated to provide structural association with new Pitkin Marsh lily colonies (section 4.2.3., 5.2.9.), or to revegetate areas cleared of non-native vegetation. Seed propagation is highly feasible in outdoor seedling beds (semi-shade, moist milled
sphagnum moss seed bed), but seedling growth of this species can be very slow for several years, and thus would require many years of nursery space to achieve adequate transplant-size specimens with high survivorship potential. Basal “sucker” shoots and rooted shoots formed by natural or artificial layering of western azalea are able to form faster-growing clones of parent plants. If seedlings are used, they should not be transplanted to the field until their roots fill are in gallon containers (or shallower containers for their fine, spreading root systems). Transplants should be made when soil is moist and plants are dormant in early winter. Transplant site selection should be limited to seasonally wet soils on site in partial shade. If transplants are made in early winter, and rainfall is average or above average, no irrigation should be necessary for establishment in the semi-shaded riparian ecotone. If transplants show signs of moisture stress or wilting by mid-summer, they should be pruned and irrigated, or covered with fiberglass mesh screen-covered wire mesh cages to reduce moisture stress.

5.2.13. Potentilla sp., (nominal P. hickmanii, Hickman’s cinquefoil

Potentilla hickmanii is a federally listed endangered species, rarely occurring in vernal wet meadows or coastal scrub on the north and central Central Coast region (Errter 1993a) has been reported from Cunningham Marsh, based on an 1880 collection by J.W. Congdon and 1946-7 collections by J.T. Howell and Milo Baker. (Errter 1993b, Best et al. 1996). Errter (1993b) has re-examined the Cunningham specimens. She found that they differed from typical P. hickmanii in key leaf characters, but were “almost a dead ringer” for one 1929 collection of P. millefolia from near Deer Creek, Oregon. Multivariate analysis of the Cunningham specimens indicated they were most similar to a more geographically remote but related species, P. plattensis (Errter 1993b). Until contemporary molecular data confirm the genetic relationships of the Cunningham Potentilla, it should not be assumed to be related to the nearest typical living populations of P. hickmanii from Montara Mountain, San Mateo Co.

The precise locality of the historic Cunningham Marsh Potentilla is unknown, and it is doubtful that it specifically occurred on the CDFG easement site. If the Cunningham Potentilla is rediscovered, or recruited from a (hopeful) dormant persistent seed bank, it should be protected in place, and promptly propagated by seed (if viable seed are produced), or by clonal divisions if plant(s) are sufficiently robust for “surgery”. Propagated replicate clones plants should be cultivated and used for:

1) cultivated conservation populations in botanical gardens with research programs, presumably including University of California at Berkeley/Jepson Herbarium (the institutional home of the leading expert on taxonomy of Potentilla, Barbara Errter);

2) experimental reintroduction to prepared, protected sites within Cunningham Marsh, especially plots near the point of rediscovery. Replicate clones should be used
as “phytometers” to probe for microenvironments suitable for adequate growth and reproduction. Because of the uncertain taxonomic affinity of the Cunningham Potentilla, I strongly recommend that no nominal “reintroduction” of P. hickmanii from offsite populations be approved or considered. All propagation, cultivation, and reintroduction plans should be developed in coordination with the California Department of Fish and Game, U.S. Fish and Wildlife Service, and at least one expert in the taxonomy of Potentilla, as well as experts in reintroduction of rare marsh plants.

5.2.14. Rhynchospora spp. (beaked-rushes)

Rhynchospora californica was historically present at Cunningham Marsh, but information on its current status or location is unavailable. Like Carex albida, Rhynchospora spp. are typically found in very acid bog-like peaty substrates with very low fertility. These habitats are present at Pitkin Marsh, but are not currently available at the CDFG easement site. Thus, the feasibility for conserving Rhynchospora spp. there are low, but possible if specific marsh sub-habitats are constructed, as discussed for C. albida (section 5.2.2.). To protect the biogeographic integrity of the Pitkin-Cunningham-Perry Marsh complex, introduction of Rhynchospora capitellata and R. alba should not be considered unless their protection or viability at the Pitkin Marsh is in severe jeopardy.

6.0 Community-specific management

6.1. Sedge-rush emergent seasonal freshwater marsh

A seasonal sedge-rush marsh remnant occurs below the concrete culvert in the tributary stream. It appears to have developed over an accreted floodplain deposit of fine sand and organic matter in recent historic time. Marsh vegetation is predominantly native (Juncus effusus, minor cover of Scirpus microcarpus, Oenanthe sarmentosa, other spp.), with some invasion by bull thistle (Cirsium vulgare). The marsh is truncated by a clonal colony of European white poplar (Populus alba) at its downstream end. It is likely that marsh elevations have been increased by sedimentation, and sedimentation has itself probably been increased by historic land uses (agriculture).

A minimum recommended marsh enhancement regime would include (a) manual removal of bull thistle rosettes in fall, winter, or spring; (b) removal of the white poplar thicket by cut-stump application of herbicide in late spring or early summer. The poplar thicket is an attractive nuisance, providing choice cover for deer that browse local sensitive plant populations like Pitkin Marsh lily. The marsh habitat would be improved and diversified by excavating some depressions and troughs aligned with streamflow, emulating abandoned blind channels that retain water or moisture later in the season, and provide habitat for a wider range of emergent marsh plants or floating plants like Lemna spp. They could also
enhance habitat for dabbling ducks seeking sheltered sites for nesting, feeding, or broods. Depressions with sloping edges and depths up to 1 m below surface substrates would be appropriate. Small manually dug wetland depressions, or backhoes may be used to create larger depressions. Backhoe and similar heavy equipment would be constrained by access through intact native vegetation, but ample recovery of vegetation is expected if disturbance occurs in late fall. Spoil disposal locations could occur in areas dominated by Himalayan blackberry. Spoil piles should be planted densely by native clonal species tolerant of wetland soils, such as *Leymus triticoides* and *Carex barbara*, to pre-empt rapid invasion by weeds.

The most suitable initial location for constructing experimental wetland depressions would be in the killed stand of white poplar (Section 5.1.5.), where no existing native plant communities would be sacrificed. Passive revegetation may be attempted and monitored, but higher initial native species richness would be achieved by introducing either propagules (seeds, vegetative fragments) or transplants of native wetland plants from nearby sources or nursery production. Isolation of the site would naturally favor dominance by widespread species with high dispersal and colonizing ability, such as cattail or willow. It is not known whether vegetation other than that already present would be stable or persistent in constructed depressions. With unknown contemporary sediment loads, the persistence of the depressions themselves cannot be predicted, but there are no indicators of recent sediment deposition locally except within the channel itself.

### 6.2. Willow riparian scrub thicket

The willow riparian scrub thicket dominates most of the channel and floodplain zone of the tributary stream on site. Few modifications are needed or feasible in this nearly monotypic vegetation. Removal (felling, cut-stump herbicide treatment) of a few immature blue gum trees (*Eucalyptus globulus*) at the upstream end of the site is warranted. Canopy gap creation near lily populations is treated in Section 4.2.4.

### 6.3 Blackberry-rush wetland transition zone

The primary vegetation management need in the riparian wetland ecotone above the willow thicket is removal of non-native Himalayan blackberry (Section 5.1.6), which is abundant and has the potential to increase dominance over time. Native dominants in this ecotone appear to be *Juncus effusus*, native California blackberry (*Rubus ursinus*), and sedges, with many other native species frequent, often near mature oaks. This vegetation is the matrix in which most Pitkin Marsh lily colonies occur. Velvetgrass (*Holcus lanatus*) is locally common to abundant in this vegetation. Techniques combining cutting/mowing, use of soil acidifiers and “antifertilizer” sawdust amendments (high C:N organic mulch to immobilize available N) may be combined experimentally to reduce competitive ability of velvetgrass in this zone. Experimental gap creation, mowing, and herbicide application are other techniques that may be attempted to control this species where its density becomes excessive, or a potential threat
to special-status species. If herbicides are applied, only those with short-lived activity such as glyphosate should be used, and only during seasons when velvetgrass produces receptive green tissues, and non-target species have minimal receptive tissues.

6.4. Oak woodland and grassland (oak savannah).

Oak woodland and grassland are treated here as a single management unit because natural grasslands and oak woodlands are assumed to have been ecotonal to one another in pre-historic conditions, with continuous oak woodland intergrading with savannah (scattered oak sin grassland matrix), depending on natural or aboriginal fire regimes. Please refer to section 6.4.1. (grassland management). Oak dispersion patterns are currently not naturally integrated with grasslands at the site, probably because of past agricultural land use patterns. Currently, grasslands occur in uplands lacking partial or full shade of tree overstories, and in areas of partial oak shade. Blackberry and other shrub thickets, however, are often abundant under trees in the fire-excluded site, possibly due in part to seed deposition shadows from birds in trees. It is unlikely that treeless grasslands dominated the pre-agricultural landscape of the Cunningham area; oak woodland or oak savannah is a more likely natural upland plant community, and may justifiably be assumed for a restoration objective. Remnants of apparent oak savannah are evident in sheets 88–89 of Sonoma County soil maps with aerial photography dating from the 1960s (USDA 1972).

6.4.1. Upland (arid-mesic) grassland management

Main objectives for management of grasslands of the site are to (1) significantly reduce the dominance of non-native grassland species in a sustainable way, (2) enhance the diversity and abundance of local native grassland species; (3) gradually re-integrate grassland and oak savannah vegetation by long-term restoration of moderate oak densities and semi-shaded understory grassland (see section 6.4.2).

Feasible management activities meeting these objectives in the long-term will depend in part on more detailed assessments of the seasonal vegetation change of the grassland areas. Non-native grasses can mask abundant native populations that are conspicuous or evident only briefly in spring. Currently, no data (maps, surveys, field markers) on the distribution or abundance of the vernal native grassland flora are documented. Multiple seasonal survey times (vernal flora, summer graminoids) with notation of rank abundance of species would be needed to adequately characterize the true condition of the grassland for management. Patches with high densities of bulbs, native forbs, or perennial native grasses should be mapped or staked in the field so that appropriate site-specific management techniques may be applied. In addition, the identification of all grasses (particularly widespread or abundant species) should be verified. The level of detail can vary from formal vegetation mapping and quantitative sampling, to a coarse grid-system with releves (subjective, descriptive estimates of ranks of relative or absolute abundance of species). For management purposes, semi-
quantitative, coarse-scale approaches will probably suffice (as in traditional range
management), and these may be most compatible with volunteer stewardship constraints.

The basic “toolbox” for management of grasslands to reduce non-native annual grasses and
enhance native California grassland species diversity includes: burning, mowing, haying
(mowing with raking/removal of cut hay), selective herbicide application, nutrient
reduction/immobilization, seeding, and planting. Not all of these techniques are equally
feasible or appropriate for a rural residential setting: for example, although occasional burning
of grassland vegetation is often highly effective for enhancing native grassland vegetation, it
is presumably infeasible (cost, adjacent land use conflict, hazard risks, etc.) here. Some of the
basic effects of burning, such as removal of accumulated leaf litter that favors annual non-
native grasses and suppresses natives (Dyer 2002, Reynolds 2001, Facelli and Pickett 1991,
Evans and Young 1970, Weaver and Roland 1952) and reduction of the short-lived annual
grass seed bank (Meyer and Schiffman 1999), can be achieved by combinations of multiple
methods (Dyer 1993, 2002).

**Burning**

Occasional dry-season burning of grassland vegetation is arguably one of the most basic and
important physical processes that has historically sustained grassland vegetation in California,
before and after human occupation (Heady 1990). It is one of the most effective tools for
restoring native annual forbs to California grasslands (Menke 1992, Meyer and Schiffman
1999), but is subject to many land use constraints. Chief among the effects of fire are
reduction of accumulated leaf litter (thatch, mulch), mortality of fire-intolerant woody
species, rapid transformation of nutrients (including significant volatilization and net loss of
soil nutrients such as nitrogen; Menke 1992), and variable reduction in density of plants and
viable seeds. Burning for reduction of non-native grass seed banks is generally prescribed for
late flowering/early seed set of non-native grasses (maximum mortality of seeds and annual
grasses, minimum potential for regeneration in the dry season). The effects of dry-season
burning are short-lived: after a single burn, grasslands can return to pre-burn vegetation
conditions in as little as 2 growing seasons (Meyer and Schiffman 1999), but long-term
regimes of burning or burning and grazing can be useful in restoring native valley grasslands
all native grassland species respond positively to burning in the short term, compared with
grazing (Hatch *et al.* 1999).

Burning is not suitable as a primary method of grassland restoration at sites like the CDFG
easement site, but may be pursued opportunistically, and with great care. Presumably
prescribed burns would be impractical to implement because of residential development
constraints. Because of the ecological benefits of even infrequent, light burns in grasslands,
prescribed fire is still recommended for consideration.
Burn frequencies for grassland management objectives can vary. For maintenance of grasslands previously restored by grazing/haying regimes (low fuel load, reduced seed bank of annual grasses), even infrequent burning (1-3 burns/20 years) may be quite sufficient. For reduction of abundant annual grasses, more frequent burns are needed because of rapid annual grass regeneration. Because of the cost and risks of burning in residential areas, only infrequent and localized burns with large windbreaks and reduced fuel loads are feasible, if at all.

Burning must be adapted to avoid undue mortality of oak saplings or relatively small oaks with thin bark layers. Close mowing and raking to reduce ground layer fuels to minimal levels may by used to avoid injury to immature oaks if prescribed burns are performed.

Mowing and haying

Mowing, haying, and grazing are other grassland management techniques that similarly reduce above-ground biomass to varying degrees, but with variable effects on nutrient cycling and accumulation of leaf litter. Mowing and haying may be used as a partial surrogate for burning when or where opportunities for prescribed fire are precluded. In the absence of all fire, grazing, mowing/haying influences, most California valley grassland vegetation is subject to degeneration and dominance by non-native grasses and forbs, or replacement by other plant communities.

Mowing during late flowering or earliest seed ripening stages of annual grasses can, over several consecutive years, reduce the dominance of the annual grass seed bank. Perennial native plants may suffer reduced production of viable seed as a result of late spring mowing, but are more likely to regenerate after several years of mowing, providing them with a net competitive advantage. Depending on the timing of native annual seed production may be reduced by late spring mowing if native annuals are locally abundant. Additional benefits of late spring mowing include (a) reduced fire hazards for adjacent residences; and (b) inhibition of invasion of grassland by shrubs, especially Himalayan blackberry. Disadvantages of late spring mowing as a sole management technique include: (a) retention of accumulated leaf litter (thatch, mulch), inhibiting native plant seedling establishment, and favoring seedling habitat of non-native annual grasses; and (b) interference with seed production and population growth of native species.

The timing and frequency of mowing can be altered as vegetation dominance shifts away from non-native grasses: delaying mowing until after seed set of native species present (late spring/early summer) may be feasible where non-native grass abundance has been successfully reduced. Late spring/early summer mowing enables greater seed set and seedling establishment of native grasses and forbs. Mowing too early in spring, during pre-flowering to early flowering stages of non-native grasses, is likely to result in tillering (shoot branching) and significant regeneration of flowering and fruiting culms, slightly later in the season. If
significant regeneration of tillers occurs after mowing non-native annual grasses, re-mowing at a slightly lower cutting height is indicated. Multiple, close spring mowings during the first or second year of mowing treatments are recommended to deplete non-native annual seed banks. These are likely to have more impact on non-native annual grasses than native grassland species with longer-lived seed banks or perennation structures.

Mowing during “green” seed set stages of annual grasses (intermediate seed ripening) is likely to result in significant deposition of viable seed. If mowing is performed after late flowering/early seed ripening stages, raking and haying (removal) of cut grass to remove significant viable seed is justified. These developmental stages of annual grasses vary annually with weather patterns, and are not consistently related to calendar dates. They should be assessed by frequent (twice weekly in warm weather) inspection from range management specialists, restoration experts, or experts in biology of grasses, or volunteers trained by them.

Properly timed haying (mowing with subsequent raking/removal of above-ground biomass) is likely to have substantially greater advantages over mowing alone. Haying reduces the accumulation of leaf litter that may inhibit native seedling establishment, and favor annual grass regeneration. Haying also gradually “impoverishes” soil, cumulatively removing nutrients in biomass (hay and litter) that would otherwise decompose and recycle to soil nutrient pools (Wedin 1992). High soil fertility is generally not desirable for maintenance of high native species diversity in many types of grasslands (Besenyei et al. 2001, Davies et al. 1999, Wedin 1992, Tilman 1993). Because non-native annual grasses have a competitive advantage in more fertile, productive sites, long-term reduction of soil litter and fertility is likely to favor native grassland vegetation. In this respect, haying can be a partial surrogate for more of the beneficial effects of burning than mowing alone. Again, as non-native annual grass populations are reduced, mowing with haying can be deferred to late spring/early summer to enable native seed production.

Grazing

Grazing is a potentially beneficial tool for management of some grasslands, but the effects of grazing are quite dependent on the intensity and timing of grazing, and may be influenced by grassland type, species composition, and grazing animal types. Overgrazing from heavy year-round cattle stocking in confined areas is generally detrimental to grassland quality, and results in overcharged soil nutrient levels, and prevalence of weedy non-native vegetation. Grazing management for native grasslands is somewhat controversial, but for some situations, careful control of grazing intensity and timing can be used towards achieving some restoration objectives. At least temporary grazing is most likely to have net benefits for restoration of grasslands when: (a) burning is infeasible, (b) rhizomatous sod-forming grasses are naturally scarce, and (c) non-native annual grasses are dominant, and native bunchgrasses and forbs are infrequent. Some bunchgrasses other than purple needlegrass can benefit from
light grazing in coastal grasslands (Hatch et al. 1999).

Not all valley grasslands should be assumed to be naturally dominated by bunchgrasses such as purple needlegrass (*Nassella pulchra*). Particularly in alluvial grasslands of the outer coast ranges and valleys with high seasonal moisture, rhizomatous sod-forming graminoid species such as creeping wildrye (*Leymus triticoides*), and sedges (*Carex barbarae, C. praegracilis*) may be natural pre-agricultural dominants (Holstein 2001). Sod-forming grasslands are likely to be relatively lower in forb and bulb abundance and diversity, but also more resistant to invasion by non-native annual grasses and forbs. Grazing is likely to be detrimental to the re-establishment of rhizomatous grasses and sedges, except very light summer grazing. There is no information on the historic natural relative abundance of bunchgrasses versus sod-forming graminoids along sandy seasonal wetlands and riparian zones of the site vicinity. It is reasonable to interpret that sod-forming, rhizomatous graminoids were a significant component of grasslands in soils moist in spring to early summer. If these are re-established, only light, infrequent grazing is indicated. *Leymus triticoides* regenerates well within the same growing season after mowing or single episodes of early summer to mid-summer grazing where soil moisture is moderate (P. Baye, pers. observ.).

If the primary purpose of grazing is to reduce biomass, competition, and seed production of dominant non-native annual grasses in a native grassland restoration program, short-rotation, intensive late spring grazing during late flowering/early seed set of annual grasses, or even wet-season grazing, is desirable (Dyer 1993). Summer grazing combined with burning can also increase abundance of native forbs (Dyer 1993), but positive interactions between combined grazing and burning are not always indicated (Fehmi and Bartolome 2003). If short-term, intensive spring grazing causes reduction in the abundance of native annuals, it would be prudent to adaptively suspend grazing for one or more years before resuming. Summer or fall grazing by sheep (which can often tolerate unpalatable, low-nutrition straw for brief periods) may be effective at reduction of litter and biomass, but is unlikely to reduce non-native annual grass seed banks.

**Herbicides**

Grass-specific herbicides are unlikely to be beneficial where mixed native and non-native grassland species co-occur; non-target impacts on native grasses and forbs are likely to be excessive and counterproductive. In areas where non-native grasses are overwhelmingly dominant, non-selective low-toxicity, non-persistent herbicides such as glyphosate may be used locally to create bare patches for subsequent seeding. Low-toxicity herbicides with potential for significant residual activity in soil (e.g. imazapyr) are not recommended. Glyphosate-treated, killed patches should be raked to clear persistent litter, so that they become suitable for seeding with native species (see below). Non-native grassland vegetation adjacent to herbicide-treated patches should always be mown during late flowering stages before herbicide application to minimize non-native grass seed dispersal into herbicide-
cleared patches.

Seeding native grass and forb species into grassland is a potentially potent method of augmenting or restoring native species abundance and diversity in grassland communities that are deficient in native seed banks, or subject to dispersal limitation from source populations (Seabloom et al. 2003). Seed addition of native herbaceous plants is most likely to be effective on relatively unproductive, low-fertility grassland soils with low abundance of annual non-native grasses. Seeding native grassland species at high densities after reducing leaf litter and competition (by haying, brief episodes of intensive grazing, burning) may result in persistent populations (Seabloom et al. 2003).

**Recommended combined methods for the CDFG easement site**

Given the difficulty of authorizing, funding, and implementing controlled burns, burning is likely to play a very limited role in grassland management of the site, regardless of its desirability for grassland vegetation management. Factors that may increase the feasibility of limited use of controlled burns include: (a) periodic (annual) to occasional (2-3 year cycle) close-cropped early summer mowing and haying to minimize fuel loads; (b) conducting only occasional (3-7 year cycle or longer) controlled burns in cool, moist late fall weather when fuel moisture is low enough for ignition, but high enough to slow burning rates; (c) use of patch-burns confined to discrete plots in metal burn-boxes.

Initial treatment of grassland vegetation should probably emphasize mowing and haying for at least several years to reduce accumulated leaf litter and non-native annual grass seed banks. Close cropping (3-4 inches or less) during late flowering stages of annual grasses, and thorough harvest of biomass, is desirable for initial conditioning. For the first two years of mowing/haying treatment, multiple spring mowings with low cutting heights (3-4 inches) are recommended. After relative abundance of annual grasses has declined, mowing height can be increased to 6 inches, and frequency and timing can be shifted to greater cutting height and later cutting time (early summer), to allow native annuals to set and disperse seed. Manual raking may be justified (volunteer work) where annual grass density is very high and native species density is very low.

Seed addition, bulb planting, and planting plugs or divisions of perennial native grasses may be indicated if (or where) reduction of non-native grasses is achieved by haying. Stock of local populations for sowing or planting should be developed by cultivating locally sampled populations prior to mowing/haying treatments, either at offsite nurseries or on site. A small (<1/10 acre) on-site “nursery” area (not developed with infrastructure) could be adequate for supplying propagules of native grasses for incremental seeding and transplanting. Propagation may be coordinated with horticulturally inclined CNPS volunteers with capacity to grow selected species offsite, and harvest and store local seed or other propagules. Seed sowing should be conducted in fall as soon as germinating (sustained soil-wetting) rains are forecast,
to minimize duration of seed predation time, and promote competitive advantage of early germination. Bulb planting should occur in cool, dry soil in fall. Plugs of bunchgrasses or rhizomatous sedges and grasses should be transplanted only in early winter when soil is moist, to maximize duration of wet-season root growth before rapid elongation of leaves in late winter.

6.4.2. Oak regeneration and oak subcanopy grassland

Native white oaks (Quercus lobata, Q. garryana, or possibly intermediates) have been observed as low seedlings in upland grasslands, and as saplings within deer-exclosures of lily populations. Juvenile oaks are otherwise rather scarce on the site, suggesting that oaks are recruiting from seed, but are limited by excessive herbivory. The same is likely to be true of Q. agrifolia and Q. kelloggii on site, since mature specimens are also producing seed. Increasing the distribution and abundance of native oaks in the grassland is a reasonable restoration goal for the site, especially in areas where non-native pines are prescribed for conversion to snags (see 6.5), but also in cleared historic pasture/orchard grasslands.

Because conservation priorities for rare endemic lily conservation should take precedence over general oak conservation, lily exclosures that are recruiting oak saplings (especially at high and unsustainable densities) may be used as a borrow source for on-site transplants of seedlings and saplings. If additional oaks are needed, they may be propagated by seed (acorn) by sowing viable acorns in outdoor, on-site seedling nurseries protected by deer exclosures. General seed propagation methods for oaks are provided by McCreary (2001) and Holmes (1996), and may be adapted to local conditions.

Seed propagation of oaks can be based on very little work: collection of local acorns in fall directly from trees (or soon afterward to prevent desiccation and loss of viability), and directly sowing in raked/prepared soil, covering them with 3 – 5 cm of soil. Direct oak seeding is reported to result in only about 5% survivorship (B. Young, CNPS Milo Baker Chapter, pers. comm.). Covering acorn plots with fine-mesh rectangular cage to restrict rodent depredation is recommended. Direct seeding of oaks into grassland at this site is not recommended because of conflicts with mowing/raking during the first 2 to 3 years of management. Similarly, transplanting of saplings into grasslands should be phased to minimize conflicts between mowing and oak establishment. Mowing around a small number of first-generation oak transplants would be feasible, but frequent juvenile oaks would be problematic.

General practical methods of oak transplanting and aftercare are found in McCreary (2001), and are adapted here. Transplanting oak saplings should involve two years of preparation: one year of pruning, and one for transplanting. Sapling taproots should be root-pruned manually with a tile spade, to a depth of approximately 0.5 to 0.7 meter to force enough lateral branching to enable survival of oak seedling transplants in the subsequent year. Tops should be pruned to a single leader shoot to encourage early attainment of transplant height.
above browsing levels. Root-pruning may be performed from late fall (after moistening to 0.5 m depth) to early spring. Top-pruning should be performed during dormancy, before bud break. Transplants should be excavated in late fall or early winter after the first post-pruning growing season, and transplanted bare-root on the same day (or following day) as excavation, to ensure that roots remain moist. Transplanting should be performed in cool, moist weather. Minimal fertilizer or none should be used on transplants, and should be applied only at the bottom of the planting hole. No soil amendments should be used. Transplants should be watered immediately after moist soil is moderately well compacted around roots, but no supplemental irrigation is recommended for the following growing season (unless winter drought occurs and soil becomes dry). Slight terracing or berming of the transplant base may be constructed to facilitate infiltration of rainfall, but no depressions should occur around the base. Transplants should be protected by staked deer fence cages. Short Treeshelters may be used temporarily around the base of the transplanted oak to protect against rodent or hare damage, but whole-trunk treeshelters may have unwanted unnatural indirect persistent effects on structural attributes (branching, mechanical strength) of the trunk. The bases of transplants should be thickly mulched (to 4” depth) with decayed wood from fallen logs or translocated tree litter, to reduce water competition with non-native grasses.

The partial shade and wind-sheltering provided by dead pines may facilitate establishment of transplanted oaks by reducing desiccation and temperature stress. Oak seedlings and smaller saplings transplanted to open grassland may also benefit from very light shading provided by either loose brush “teepees” around deer exclosure cages, or fiberglass mosquito screening fastened to the top and sides of cages. Fiberglass screens or brush can be removed after the second year of establishment.

Density and spacing of oaks should be irregular, leaving large gaps of open grassland between clusters of oaks. Nearest neighbor distances between oaks should be a minimum of about 20 feet.

The composition of grassland under oaks would naturally vary somewhat from open grassland. *Elymus glaucus*, terrestrial *Carex* spp. *Luzula*, and bulbs (*Tritelia*, *Brodiaeae*, *Calochortus*) are often found at relatively higher densities in partial woodland shade. Otherwise, grassland management recommendations from section 6.4.1. apply to oak understory grasslands.

Transplants of oak seedlings to grassland or pine/grassland should also be performed in fall. Similar root-pruning one year in advance of transplanting is also recommended (to depth of about 0.5 m). Oak seedlings found in the grassland should be caged with staked deer fencing until they reach a height above browsing level.
6.5 Monterey Pine stand

The large introduced Monterey pines between the grassland and riparian willow thicket are very large, old and decadent, and some have become decadent snags (dead standing trees) with abundant woodpecker cavities. The pines do not appear to be highly invasive, but some juvenile pines have established, and may maintain or possibly increase the density of the population in the long term. The pines as a living population have negative value for native plant community conservation value, but as dead standing trees (snags), they have very high wildlife habitat value for cavity-nesting owls, woodpeckers, and bats. Fallen limbs and trunks are providing downed woody debris. Large snags are otherwise scarce in suburban developments or agricultural lands because dying or diseased trees are removed before they can become safety hazards or esthetic nuisances. Within the CDFG easement site, however, snags are unlikely to be hazards to anyone, and against the oaks and riparian thickets, they are probably not esthetically objectionable to neighbors. Residential land uses are compatible with local populations of owls and bats that reduce populations of rodents and mosquitoes (respectively). The canopy space occupied by pines competes with potential native oak space. The weak shade provided by snags, and partial browsing deterrence provided by fallen pine branches and limbs, may facilitate establishment of naturally recruited or planted oak seedlings or samplings.

Juvenile or young mature (<6” DBH) Monterey pines should be manually removed with a bow saw or chain saw. Felling small trees in fall or winter would minimally disturb regeneration of scarce native grassland species. Small felled trees can be left in place, or slash and wood can be disposed on-site within any of the large Himalayan blackberry thickets. If acceptable to neighbors, converting some (or all) of the most senescent large trees to snags can be achieved gradually by girdling the base of the tree with an axe, removing the cambium layer below the bark, and exposing at least 4” of wood. Trees girdled in fall would be likely to die within the next growing season. Partial girdling (leaving a few wounded inner bark strips as bridges across the girdle) can be used to stagger or phase the rate of tree mortality over one or two seasons.

Girdling and snag conversion are significantly less expensive methods of removing exotic pines than professional felling and removal, and provide important, scarce tree cavity habitat for wildlife. Cultivation of snags would probably increase existing populations of acorn and hairy woodpeckers, bats, and owls. Girdling is slow and silent, and requires no special equipment, and generates no loud noise. Felling large pines, in contrast, would also require truck access for removing sections of trunks, and would cause acute noise nuisances for adjacent landowners. Snag cultivation could be implemented relatively quickly, and thus facilitate prompt initiation of oak regeneration in the area occupied by pines.
6.6. Cattle grazing buffer zone

The intensively grazed and manured ranchland adjacent to the CDFG easement site, near the sedge-rush marsh patch below the culvert, is a probable source of nutrient-enriched runoff and wind-dispersed weed seeds. There may not be a feasible way to mitigate nutrient—enriched groundwater in baseflows to the creek, but surface runoff from the manured soils and wind-dispersed weed seeds may be intercepted to some extent by planting a multi-story shrub and tree “buffer” at the edge of the site’s border with the ranchland. Existing grassland has limited capacity to assimilate nutrients or retain them in surface litter. Planting a mixture of local oaks, native blackberry, California wax-myrtle and even fast-growing coyote-brush (Baccharis pilularis) as a buffer would probably be better than leaving the pasture edge in open herbaceous vegetation. The existing vegetation along the border appears to be dominated by weedy forbs, grasses, and Himalayan blackberry. If nutrient assimilation rates are to be maximized in the buffer zone, trees and shrubs in the buffer zone should be periodically coppiced (severely pruned to force rapid, productive regrowth) after they are well-established.

7.0. Implementation schedule for vegetation management.

The following schedule summarizes all suggested general vegetation management tasks. Years 1-5 are based on growing seasons, beginning with fall rains. Beyond year 5, vegetation management would consist of adaptive re-application of methods as needed. Grassland management would be required annually, but all other active vegetation management actions (other than monitoring) should decrease over time.

YEAR 1.

1.1. Fall: perform willow canopy thinning around overtopped lily colonies.
1.2. Fall: collect and field-sow local acorns of all oak species in outdoor beds for future use.
1.3. Fall: Root-prune oak saplings targeted for transplanting in year 2.
1.4. Fall: Search and cage (exclosures) established oak seedlings in grassland (deer protection).
1.5. Winter: inspect site for zones of persistent soil saturation, ephemeral streams, inundation. Measure water depth in stream, limits of high water lines after storms. Mark in field. (Future reference points for vegetation management)
1.6. Winter: set up large grid system (keyed to fence line) to assist plot locations for plant populations and vegetation management activities.
1.7. Winter: Plant buffer zone along cattle-grazed adjacent parcel opposite rush-sedge marsh below concrete culvert.
1.8. Early winter: Transplant field-collected divisions of selected native perennial grasses to a small on-site “nursery” to use as source of propagules for seeding, division.

1.9. Late winter: inspect for vernal flora. Flag, stake or map locations of significant populations for future management (enhancement, propagation, etc.)

YEAR 2.

2.1. Spring: perform mowing at/slightly after peak flowering of non-native grasses. Repeat if tillering and re-flowering occurs.

2.2. Remove hay after drying.


2.4. Fall: perform lily seed collection; initiate ex situ clonal bank propagation, in coordination with CDFG and USFWS.

2.5. Fall: initiate vegetation gaps/litter removal plots for lily seed sowing and seedling recruitment.

2.6. Fall: Lift and transplant Year 1 root-pruned oak saplings and distribute to grassland or riparian ecotone transplant sites.

2.7. Fall: sow lily seeds in outdoor propagation beds.

2.8. (All seasons) search and record locations, describe population status of plant species of concern, or native species that may be used as stock for revegetation of disturbed gaps or seeding.

2.9. (Summer and fall) Monitor lily clone size, reproductive output, end-of-season seedling survivorship.

YEAR 3

3.1. Spring: repeat mowing/haying schedule focused on reducing non-native annual grass seed production where annual grasses remain abundant. If annual grasses are depleted in large patches, delay mowing until after seed set of native grassland vegetation.

3.2. Monitor year 2 experimental vegetation treatments; continue additional field trials as indicated by initial results.

3.3. Continue iterations of blackberry patch removal by feasible methods (mechanical, herbicide); revegetate in wake of removal operations in fall.

3.4. Maintain 1st year lily seedling plants (expect 2-3 years to achieve clonal transplant size, 4-5 years for flowering size in optimal cultivated conditions)

3.5. Fall: sow native grassland species seed and transplant native grassland vegetative propagules into successfully treated patches of grassland with minimal 

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annual grass reproduction, to increase native species diversity.

3.6: Fall: search and cage additional oak seedlings

YEARS 4 -5

Ongoing adaptive maintenance. Annual summer mow grasslands; delay raking to maximize seed dispersal of native grasses and forbs. Spring-mow outbreaks of annual grasses. Maintain canopy gaps in riparian zone for lily populations as needed.
Table 1. Selected native wetland plant species historically collected or reported from Cunningham Marsh or vicinity, and related wetlands of southern Sonoma County (Pitkin, Perry Marshes). Source: Flora of Sonoma County (includes Rubtzoff 1953), Best et al. 1996. Nomenclature and taxonomy follow Hickman 1993 unless otherwise noted. Species narrowly associated with oligotrophic wetlands (acid sandy or peaty bog-like marsh, wet meadow, swamp, fen, carr, or true bog) are indicated by a square symbol [□]. Regionally rare plants are indicated by ®. Localities reported [x] are approximate historic descriptions, not known boundaries.

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“Laguna east of Cunningham”
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<td>Rhynchospora alba</td>
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<tr>
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<tr>
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<td>&quot;Laguna north of Cunningham&quot;</td>
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<td>Scirpus cernuus</td>
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<td>Vaccinium cespitosum</td>
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Figure 1. Soil Survey map of Sonoma County, Sheet 96 (USDA Soil Conservation Service 1972), showing unnamed tributary of Blucher Creek (syn. Bloucher Creek) northeast of “Knowles Corner”, including CDFG easement site. Soil series codes: GdC: Goldridge fine sandy loam, 2 to 9 percent slopes; GdD: Goldridge fine sandy loam, 9 to 15 percent slopes; GdE: Goldridge fine sandy loam, 15 to 30 percent slopes; BcA: Blucher loam, 0 to 2 percent slopes. Sb_ : Sebastopol series (similar to Goldridge)
Figure 2. Cropland and pasture with partial conversion to residential land use surrounding project site at Cunningham, April 2005.
Figure 3. Predominantly non-native grassland (former pasture?) with infrequent oak seedlings on easement site. April 2005.
Figure 4. Monterey pines (former windbreak plantings?) near edge of riparian woodland within easement area; snags and decadent mature trees (majority), and juveniles. April 2005.

Figure 5. Riparian ecotone: transition between riparian wetlands (willow thicket, sedge-rush marsh) with grassland, scrub, oak woodland. April 2005
Figure 6. Riparian wetland zone of the site: rush-sedge and water-parsley marsh, blackberry and azalea thicket, and willow thickets, in soils saturated or flooded in winter and spring, mesic in summer. April 2005.
Figure 7. Staked fencing for exclusion of deer browsing around lily populations on site. April 2005. (a) tomato-cage for isolated young plant; (b) small rectangular “cage fence” with vineyard stakes, isolated small clonal population; (c) large enclosure with multiple clonal populations among azaleas.
Figure 8. Lily clones emerging under and through western azalea “nurse shrubs”.

Map of CDFG Easement
References


Kyser, G.B. and J.M. DiTomaso. 2002. Instability in a grassland community after the control of yellow starthistle (Centaurea solstitialis) with prescribed burning. Weed Science 51:


